2 PRESENT STATUS AND HISTORICAL TRENDS IN WATER QUALITY

Overflow pipe on North Mill Pond

The ability of an estuary to support a variety of unique habitats, diverse assemblages of organisms and a variety of human activities is largely dependent on environmental quality. Waters that can affect estuarine water quality include groundwater, precipitation, wetlands and surface waters, including estuaries, rivers, lakes, streams and ocean waters. Water quality in turn is dependent on the types and amounts of contaminants that enter estuaries as a result of human activities, and the natural processes of an estuary that transform, assimilate and transport contaminants. Both humans and natural ecosystems depend on certain levels of water quality for providing safe drinking water and as habitat for sustained food sources. There are many other human uses of the estuary and its surrounding environment, some of which may contribute to contaminant loading. The following chapter is organized by contaminant category in order to summarize information for each category, to frame issues, to assess the significance of issues and to develop the context to formulate corrective management strategies where necessary. Generally speaking, the primary contaminants of concern for most estuaries, including those in New Hampshire, are:

1 microorganisms from improperly treated sewage, urban stormwater runoff and other nonpoint sources;

2 nutrients from point sources (sewage treatment plants) and nonpoint sources (riverine input, surface runoff, septic systems, atmospheric deposition, etc.);

3 toxic contaminants (trace metals, organics, oil, pesticides, etc.) whose sources may be historic (chromium, pesticides), potential (oil) or current (metals and PAH's from stormwater, industrial and municipal wastewater and atmospheric deposition);

4 sediments of upland watershed or riparian origin that are carried into the estuaries by runoff.

These contaminants are listed in no particular order of priority. This section of the report describes the current status and spatial and temporal trends of these contaminants in coastal New Hampshire, and provide information on documented and suspected sources. Documented and potential impacts to living resources are also discussed. The term 'contaminant" is used most often because the alternative term, 'pollutant', is only used when there are biological effects associated with the presence of chemicals in the environment.

OVERALL WATER QUALITY AND USE SUPPORT

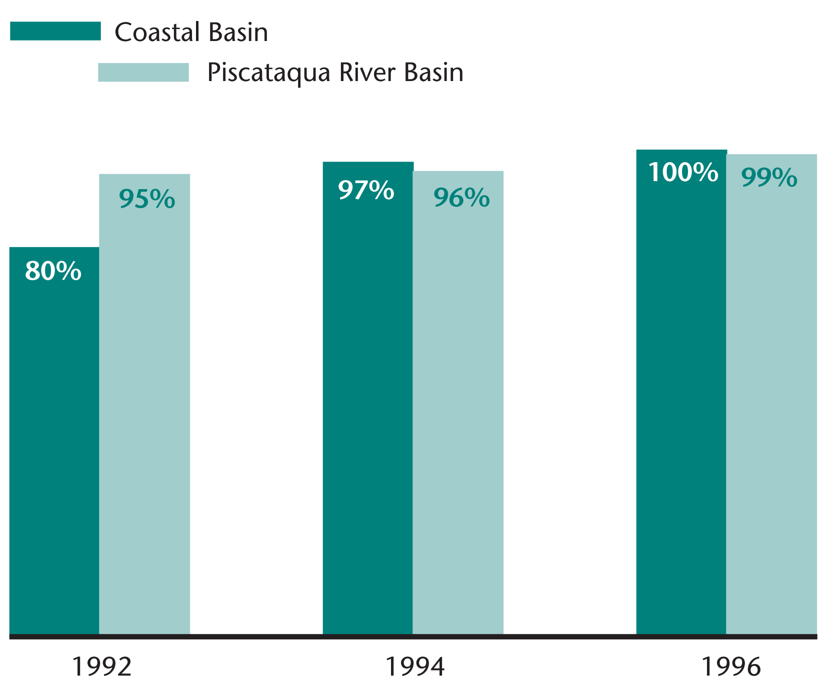
2.1.1 BACKGROUND

The Federal Water Pollution Control Act, as reauthorized by the Water Quality Act of 1987, requires New Hampshire to submit a report that describes the status of ground and surface waters to the US Environmental Protection Agency (USEPA) and Congress every two years. These "305(b)" reports have been published every two years since 1988. Surface waters are assessed according to overall quality and use support, individual use impairments, causes of impairments, trends in water quality, wetlands and public health/aquatic life concerns. More detailed summaries of overall quality/use support and some individual use impairments are summarized in Appendix F for the 1988 through 1996 305(b) reports.

Overall water quality and use support data are separated into freshwater and tidal waters, then by defined areas in the coastal area. The classification for use support provides information on the miles of freshwater streams and rivers in the Coastal and Piscataqua River basins supporting all uses. The tidal waters include the open ocean (Isles of Shoals), coastal shoreline and the estuaries as separate areas. Figures 2.1 and 2.2 summarize the trends in water quality for these waters from 1992 to 1996. Water bodies are classified as either "fully supporting", "partially supporting" or "not supporting" all uses. The definitions for these classification categories are as follows:

- fully supporting: criteria for contaminants or conditions are not exceeded, or are exceeded infrequently for any measurement, and no bans/advisories are in effect;
- partially supporting: criteria for contaminant exceeded at low to medium frequency for any measurements, restricted consumption advisory or ban in effect, or advisory lasting only a short period;
- not supporting: criteria exceeded at medium frequency, advisory periods too long or too frequent, or "no consumption" ban in effect.

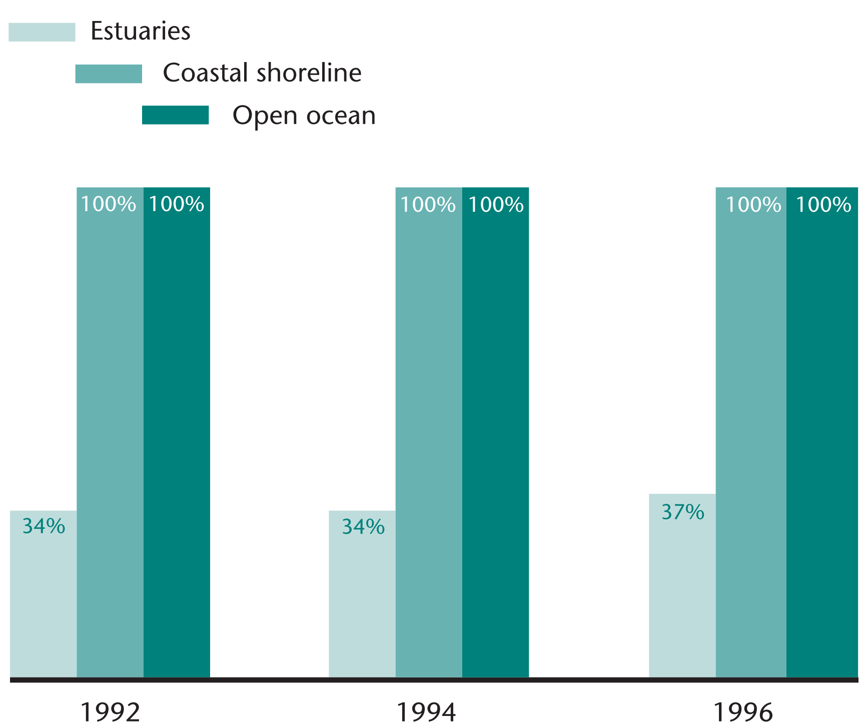
FIGURE 2.1 *Percent of classified coastal waters as fully supporting all uses: Freshwater (NHDES, 1996b).*



These classification categories are defined in more detail for the different individual use categories, including aquatic life use, drinking water use, recreational use and fish consumption use, based on USEPA guidelines. The aquatic life use category criteria are based on conditions where chlorine, ammonia or other toxicants cause violations based on acute toxicity tests, or conditions relative to dissolved oxygen, pH or temperature exceed criteria limits.

The overall program of assessing water quality and use support has evolved since 1988. In general, less information was available in earlier years for assessing surface waters, and the assessment of some uses was incomplete. More recent data, showing a high degree of support for all uses, are more complete and therefore more accurate relative to a greater range of contaminants. Between the 1990 and the 1992 305(b) reports, the USEPA suggested that New Hampshire and other states use a new database (Waterbody System software; River Reach File-RF3) for defining hydrologic features. The miles for surface waters reported by New Hampshire decreased from 14,544 to 10,841 miles as a result of differences in scale used to trace hydrologic features. In previous years, NHDES only assessed, or made use support decisions, on 1348 miles statewide. The assessed waters tended to be "problem" waters. In 1992 and thereafter, NHDES has used any available information to assess all waters, and area/mileage assessed for all freshwater and estuarine waters thus increased from 1990 to 1992. Other changes in the program resulted from passage of HB 560, amending RSA 485:A, by the legislature in 1991. Thereafter, all existing Class C waters were reclassified and upgraded to Class B, with the goal of attaining "fishable and swimmable" conditions in all surface waters. HB 560 also included adoption of different bacterial indicators for freshwater and tidal waters. Based on EPA recommendations, fecal indicators were changed as *Escherichia coli* was adopted for freshwater and enterococci was adopted for tidal recreational waters. RSA 485:A was also changed to allow for use of any indicator adopted by the National Shellfish Sanitation Program (NSSP) for classification of shellfish growing waters.

Percent of classified coastal waters as fully supporting all uses: Tidal water (NHDES, 1996b). **FIGURE 2.2**



Hampton Beach

2.1.2 STATUS AND TRENDS OF OVERALL WATER QUALITY AND USE SUPPORT

There has been a general improvement in water quality in the fresh and tidal surface waters of New Hampshire since 1988 that can be attributed in large part to improvements in sewage treatment facilities. In the Coastal Basin, at least 75% of the rivers and streams have fully supported all uses since 1988, improving to 100% support of all uses in 1996 (Figure 2.1; NHDES, 1996b). The Piscataqua River Basin has had as little as 45% of rivers and streams supporting all uses (NHDES, 1990). In 1996, only 11 of 1001 miles of freshwater rivers and streams in the Piscataqua River Basin were partially or not supporting full use.

For all uses of New Hampshire's open ocean and coastal shoreline areas, only swimming restrictions were impairments from 1992 to 1996. This areas has since had shellfish harvesting closures imposed. From 1992 to 1996, the coastal basin and open ocean waters fully supported all uses (Figure 2.2). Estuaries have had large areas with classifications that reflect impaired use because of restrictions on shellfish harvesting due to the presence of indicators of pathogens (Figure 2.2). Recent efforts to reclassify shellfish waters have resulted in improved use support in 1996. Indicators of pathogens also caused decreased support for swimming in open ocean and coastal shoreline areas from 1988-1992, while estuarine waters have had no restrictions on swimming.

Whole effluent toxicity tests decreased uses of some coastal tributaries in 1992, and the presence of elevated metal concentrations decreased use support in tidal waters in 1994. Metals also impaired use of some freshwater streams in 1996. Aquatic life support was impaired in the Lamprey River in 1994 because of metals (NHDES, 1994). Only 4.4 square miles of estuarine waters supported aquatic life

National Pollutant Discharge Elimination System (NPDES) permitted sites in coastal New Hampshire area for which monitoring data are available in the Permit Compliance System database.

TABLE 2.1

NEW HAMPSHIRE

Wastewater Treatment Plants (WWTP)		Receiving waters
NH0020966	Wallis Sands, Rye	Atlantic Ocean
NH0100196	Newmarket	Lamprey River
NH0100234	Portsmouth	Piscataqua River
NH0100251	Rollinsford	Salmon Falls River
NH0100277	Somersworth	Salmon Falls River
NH0100455	Durham	Oyster River
NH0100609	Rockingham County Complex (prison)	Ice Pond Brook
NH0100625	Hampton	Tide Mill Creek
NH0100668	Rochester	Cocheco River
NH0100676	Milton	Salmon Falls River
NH0100692	Epping	Lamprey River
NH0100854	Farmington	Cocheco River
NH0100871	Exeter	Squamscott River
NH0101028	Star Island Conference Center	Atlantic Ocean
NH0101141	Newington	Piscataqua River
NH0101192	Newfields	Squamscott River
NH0101303	Seabrook	Atlantic Ocean
NH0101311	Dover	Piscataqua River
NHG640006	Swains Lake Village Water District	Swains Lake via wetland
Industry		
NH0000469	Tillotson Healthcare Co., Rochester	Salmon Falls River
NH0001091	KJ Quinn & Co., Inc., Seabrook	Cains Brook
NH0001490	Simplex	Piscataqua River
NH0001503	Bailey Corp.	Hunts Island Creek
NH0020923	Little Bay Lobster	Piscataqua River
NH0022306	Morton International, Seabrook	Cains Brook
NH0022055	EnviroSystems-Hampton	Taylor River
NH0022985	Aquatic Research Organisms	Taylor River
NH0090000	Pease	Piscataqua River
NHG250317	GE Somersworth	Salmon Falls River
Power Plant		
NH0001601	PSNH Newington Station	Piscataqua River
NH0001473	PSNH Schiller Station	Piscataqua River
NH0020338	Seabrook Station	Atlantic Ocean
Water Treatment Plant		
NH0000884	Portsmouth (Madbury)	Johnson Creek
NH0001031	UNH	Oyster River
NHG640007	Newmarket	Lamprey/Piscassic rivers

MAINE

Wastewater Treatment Plants (WWTP)		Receiving waters
ME0101397	Berwick Sewage District	Salmon Falls River
ME0100285	Kittery	Piscataqua River
ME0100820	South Berwick Sewer District	Salmon Falls River
Industry		
ME0000868	Portsmouth Naval Shipyard, Dry docks	Piscataqua River
ME0022861	Pratt & Whitney	Great Works River
ME0022985	Watts Fluidair, Corp., Kittery	Wilson Creek

use in 1996, the other areas only partially supported aquatic life because of elevated levels of PCBs in lobster tomalley (NHDES, 1996b). Overall, none of the estuarine water supported full use because of either PCBs or pathogens. Recreational uses and fish consumption were fully supported in all estuarine waters. The health advisory for lobster tomalley is probably the result of historical PCB contamination, and the re-classification is based on studies conducted in the late 1980s and early 1990s (Isaza et al., 1989; Schwalbe and Juchatz, 1991).

Septic systems, land disposal of solid wastes, stormwater runoff, CSOs and point sources have been the most common suspected sources cited in 305(b) reports for non-support, although the estuarine sources of the PCBs responsible for the lobster consumption advisory are unknown. The presence of pathogens, indicated by the presence of elevated concentrations of fecal indicator bacteria, has been the most common pollutant. Other problem pollutants and conditions have been in-stream toxicity, low dissolved oxygen, ammonia and metals. The trends presented in the two figures reflect to a great extent the evolving program of assessment.

The State of New Hampshire regulates point sources primarily through the National Pollutant Discharge Elimination System (NPDES). Dischargers are required to obtain discharge permits and the discharge has to meet set limits. The permitted dischargers in New Hampshire and Maine are listed in Table 2.1. Sites are categorized as wastewater treatment facilities (WWTFs), industries or power plants. There are 19 WWTFs, ten industries and three power plants permitted dischargers in coastal New Hampshire waters, and three WWTFs and three industry permittees in Maine that discharge into the waters of the Great Bay Estuary.

The NPDES program is a source for a limited range of general contaminant data in point source effluent. Monitored permit data are available from the Permit Compliance System database which is maintained by the USEPA. The NHDES and the USEPA both get reports from permittees and act on violations, should they occur. A review of data for 1996 at all permitted sites in Table 2.1 showed violations of bacterial indicator limits were frequent at some sites and were always met at other sites. Only rare violations of limits for discharges of metals occurred. Various toxicity assays are used on effluent at most facilities other than some power plants. Some facilities had no violations while others had occasional violations of toxicity limits. Two WWTFs in New Hampshire had problems with meeting ammonia discharge limits.

In general, the water quality in coastal New Hampshire has improved. The major factor has been improved sewage treatment facilities capabilities for eliminating microbial contaminants from their discharges. However, both monitoring activities and the contaminants measured have increased during the last ten years, resulting in identification of previously undocumented causes for use limitations. These changes have occurred while loading characteristics, discharge permit requirements and contaminant issues have changed to reflect evolving concerns. There is a continuing need to identify and reduce or eliminate sources of pollutants that are presently responsible for limitations on uses of the state's estuarine and coastal waters.

STATUS AND TRENDS OF MICROBIAL PATHOGENS AND FECAL INDICATORS

Humans are susceptible to diseases caused by waterborne microorganisms. Some viruses, bacteria and protozoa are human pathogens, and their presence in surface waters and shellfish is a public health threat. Some pathogenic microorganisms are present naturally in estuaries and coastal waters. The ecology of many of these indigenous microorganisms is not well understood, and their presence would be difficult to manage. However, most waterborne pathogens of concern in northern New England are of fecal origin and thus are not natural inhabitants in estuarine waters. These microbes are introduced into coastal waters largely as a result of human activities, and can thus theoretically be controlled. Known anthropogenic sources include inadequately treated wastewater discharges, septic systems, boat discharges, urban and agricultural runoff and sanitary landfills, although significant contamination can also come from waterfowl and other wildlife.

2.2.1 PATHOGENS, BACTERIAL FECAL INDICATORS AND WATER QUALITY STANDARDS

The State of New Hampshire, along with every other jurisdiction that has the need to assess water quality and classify waters, uses bacterial indicators of fecal contamination to assess the sanitary quality of water. The number of potential fecal-borne pathogens, both bacterial and viral, are too numerous and difficult to measure on a routine basis. New Hampshire presently uses fecal coliforms for shellfish growing waters, as recommended by the National Shellfish Sanitation Program (NSSP, 1995). For recreational uses of marine and estuarine waters, enterococci are used, and *Escherichia coli* is used for freshwater recreational uses, both as recommended by the U.S. EPA. The bacterial indicator standards for classifying surface waters in New Hampshire are summarized in Table 2.2. These indicator bacteria have been chosen as the best indices of fecal contamination for the different purposes based on numerous studies. In many studies conducted by UNH/JEL, *Clostridium perfringens* is also included as an indicator of long-term fecal contamination and contamination associated with resuspended sediments. The following is a summary of information on the status and trends of these indicator bacteria, with some limited information on actual bacterial pathogens and viruses. Because of the extensive amount of data for the numerous bacterial indicators that have been used, fecal coliform data will be used for most illustrations of spatial and temporal trends.

TABLE 2.2

Bacterial indicator standards for surface water classification: freshwater, tidal recreational waters and shellfish-growing waters.

Surface water	Classification	Indicator	Geometric Mean Concentration*	GMC # of samples	Maximum Limit Concentration*	MLC Frequency
Freshwater	Class A	*Escherichia coli*	47	3 in 60 days	153	1 of 3 samples
FW designated beach	Class A	*Escherichia coli*	47	3 in 60 days	88	1 of 3 samples
Freshwater	Class B	*Escherichia coli*	126	3 in 60 days	406	1 of 3 samples
FW designated beach	Class B	*Escherichia coli*	47	3 in 60 days	88	1 of 3 samples
Tidal Recreational		enterococci	35	3 in 60 days	104	1 of 3 samples
Shellfish-growing	Approved	Fecal coliforms	14	30 (most recent)	>43	<10% of samples
	Restricted	Fecal coliforms	14-88	30 (most recent)	>260	<10% of samples
	Prohibited	Fecal coliforms	>88	30 (most recent)		

* Concentrations per 100 ml

2.2.1.1 Spatial Distribution

The spatial distribution of bacterial indicators in coastal New Hampshire has been relatively well documented in most areas. Adequate spatial coverage of sampling is necessary to aid in the identification of contaminant sources and to document the effects of efforts to reduce pollution sources. In general, bacterial contaminants are present at higher concentrations in tributaries in comparison to the main estuarine waters (Great Bay; Hampton Harbor) and the Atlantic Ocean. This is a function of the most important sources of contaminants being present upstream and along the shorelines of the tributaries, the smaller volumes of water in tributaries having less capacity for favorable dilution impacts on contaminant concentrations, and contaminants are subject to physical and biological processes that remove them from water as a function of time, distance and changing environmental conditions during transport through the tributaries to the main water bodies.

Early data on bacterial contamination can be found in Jackson (1944). These data reflected the high concentration loading of untreated sewage into the tributaries to Great Bay Estuary, all of which had average total coliform concentrations of >800 /100 ml, with averages ranging from 803 to 9,020/100 ml. Concentrations were much lower at sites in Great and Little bays, but remained elevated compared to more recent data, ranging from 20 to 144/100 ml and generally in excess of the limit of 70 total coliforms/100 ml for shellfishing. In 1974, the New Hampshire Water Supply and Pollution Control Commission (NHWSPCC) reported median total coliform concentrations ranging from 50/100 ml at an upstream site in the Exeter River to 109,000/100 ml at an upstream site in the Cocheco River in freshwater tributaries (NHWSPCC, 1975). In tidal waters, concentrations were <21/100 ml at Hampton Harbor, the Atlantic coast areas and in the Bellamy River, but ranged up to 307,000/100 ml in the Cocheco River.

State agencies have conducted routine monitoring of coastal waters for over 30 years. Freshwater sites are monitored by NHDES, with NHDES, NHDHHS and NHF&G monitoring tidal waters. Citizen volunteers have also been involved in monitoring microbial water quality in the coastal waters. The Great Bay Watch has monitored fecal coliforms at up to 24 sites in the Great Bay Estuary for over ten years (Reid et al., 2000). UNH and JEL have contributed substantial water quality data as a result of numerous studies throughout coastal New Hampshire.

Great Bay and Upper Little Bay with Squamscott/Exeter and Lamprey Rivers

This area extends from the dams on the two rivers through all of Great Bay and upper Little Bay to Fox Point and the area south of the mouth of the Oyster River (Figure 2.3). The most spatially and temporally intensive database for bacterial contaminants in Great Bay is the NHDHHS shellfish water monitoring program database. The data for 12 of the NHDHHS sampling stations (Figure 2.3) were reviewed and interpreted as part of the 1995 sanitary survey for the approved shellfishing areas in Great and Little bays (NHDHHS, 1995; Jones and Langan, 1995b). Fecal coliform concentrations were low enough to support an approved classification for much of Great Bay, although elevated concentrations near the mouths of the Lamprey, Squamscott, Oyster and Winnicut rivers only supported restricted or prohibited classifications. Major rainfall events had significant negative effects on water quality throughout the area and were noted as a potential condition for classification. The area near the mouths of the Squamscott and Lamprey rivers has recently been subject to more detailed monitoring to better define the boundary between restricted and approved classifications. Dye studies for the Durham and Newmarket wastewater treatment facilities (WWTFs) plus the Great Bay Marina have been conducted, and the results will provide needed data to better define safety zones in areas

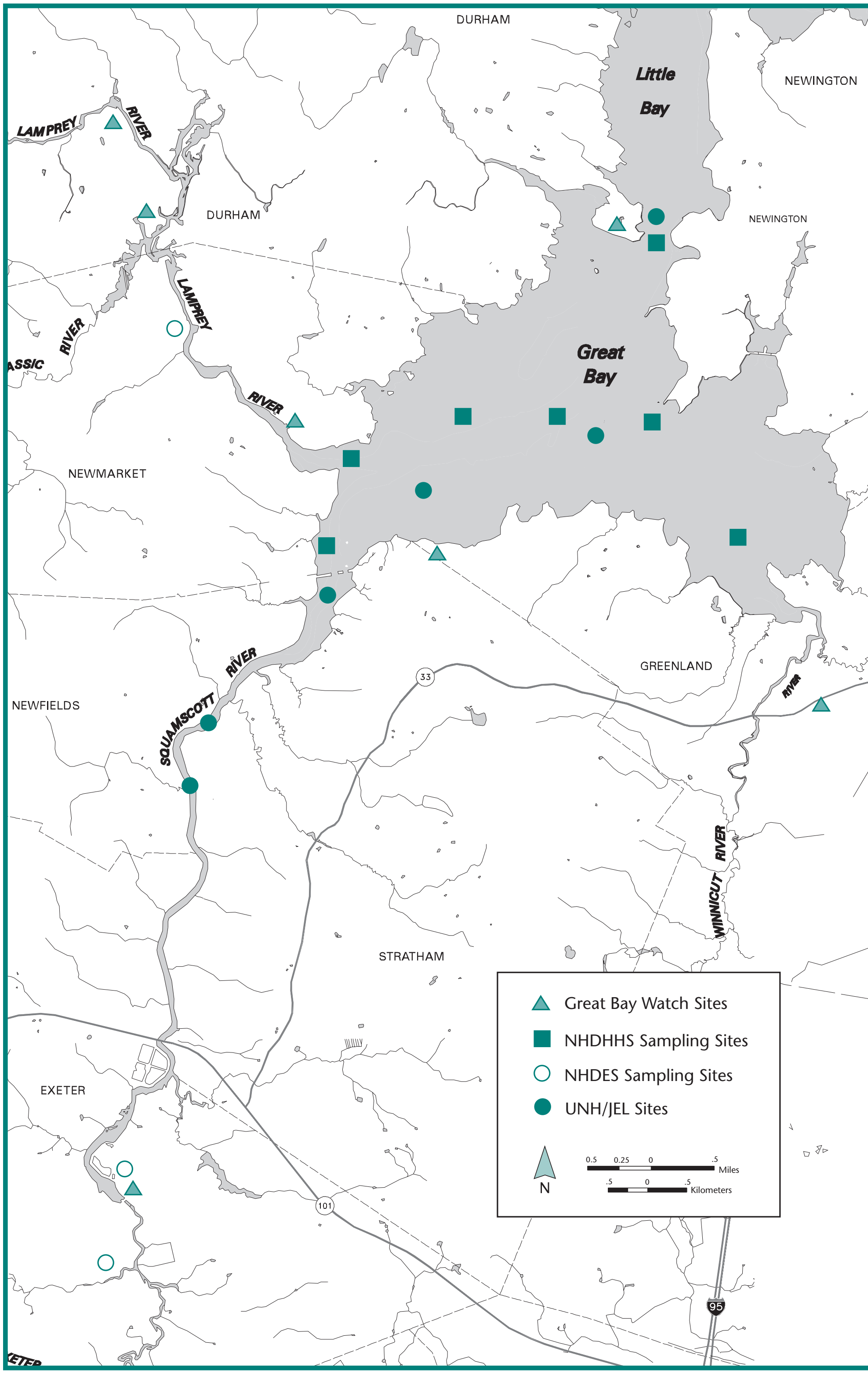


FIGURE 2.3

Great Bay, Upper Little Bay, Squamscott/Exeter River and Lamprey River water quality sampling region.

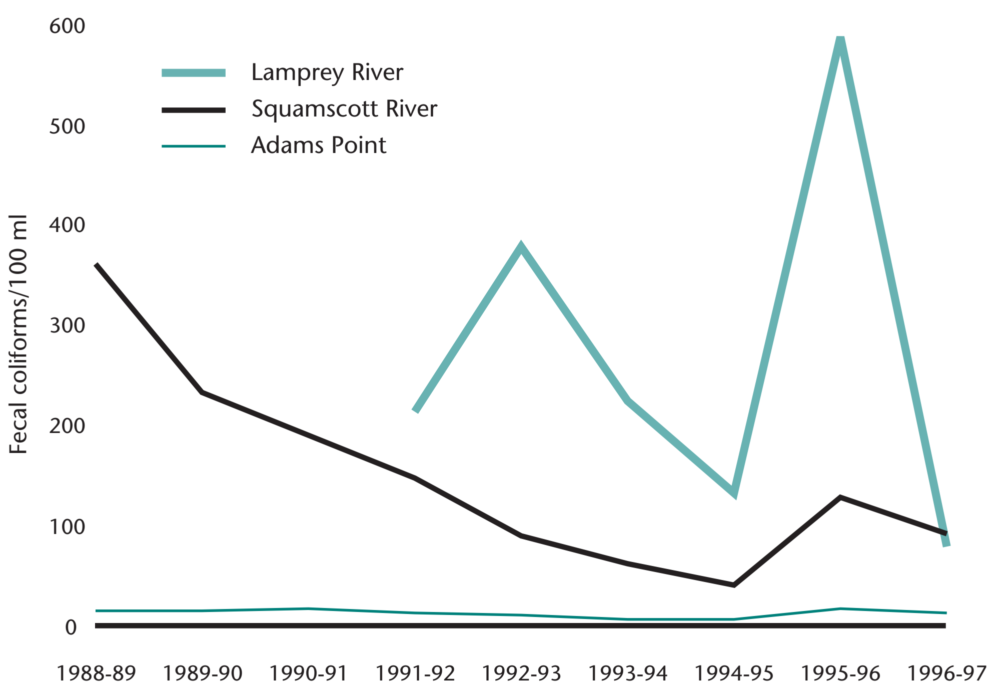
around the mouths of the Lamprey and Oyster rivers and in Little Bay.

The long-term Great Bay National Estuarine Research Reserve (GBNERR) monitoring program has provided an eleven year database for fecal coliforms, enterococci, *E. coli* and *C. perfringens* at Adams Point between Great and Little bays, Chapmans Landing in the Squamscott River and at the Town Landing on the Lamprey River (Langan and Jones, 2000; Langan and Jones, 1997). In 1996-97 as in 1988-97, fecal coliform, *E. coli*, enterococci and *C. perfringens* concentrations were lowest at Adams Point at both high and low tides (Figures 2.4 and 2.5; Appendix G). Most indicators have been present at relatively low concentrations in the Squamscott River at high tide, whereas at low tide contaminant concentrations have been much higher. The large difference in contaminants in the Squamscott River is a result of dilution with less contaminated bay water at high tide. Bacterial indicators in the Lamprey River are present at elevated concentrations at both high and low tides. Similar observations, i.e., elevated bacterial levels in the Lamprey River compared to other areas in Great Bay at both high and low tide, have been reported by the Great Bay Watch (Reid et al., 2000). The Town Landing area appears to be significantly affected by undefined localized conditions that are currently under investigation by state agencies.

The water quality in the tributaries to Great Bay has been assessed as part of numerous other studies. Both the Lamprey and Squamscott rivers were part of a three year project to investigate the effects of storm events on water quality in all tributaries (Figures 2.6 and 2.7) to the Great Bay Estuary (Jones and Langan, 1994a; 1995a; 1996a). An analysis of all three years can be found in Jones and Langan (1996a). The geometric mean fecal coliform (FC) concentrations were relatively low during dry weather over the three year study at the freshwater sites just above the dams on both the Lamprey (9 FC/100 ml) and the Squamscott (31 FC/100 ml) rivers (Figure 2-6). Compared to the freshwater sites, the concentration at the tidal water sites were lower in the Squamscott (23 FC/100 ml) and higher in the Lamprey (48 FC/100 ml) during dry weather. Concentrations increased significantly at all four sites during storm events (Figures 2.6

FIGURE 2.4

Temporal trends for fecal coliforms (colonies/100 ml) at three sites in the Great Bay Estuary at low tide.



and 2.7). During the same years, fecal coliform concentrations in the Squamscott River downstream of the dam in downtown Exeter were generally >50/100 ml (Reid et al., 2000). Fecal coliform concentrations in the Winnicut River have been elevated compared to most other sites in Great Bay at low tide, but are diluted to low concentrations at high tide (Reid et al., 1998). The small tributaries that flow into the Winnicut River and the southeast corner of Great Bay were sampled during 1994-95 (Jones and Langan, 1995b). Despite some elevated concentrations of fecal coliforms, the tributaries appeared to have little impact on water quality in Great Bay.

Both the tidal and freshwater portions of the Squamscott/Exeter River watershed were studied in detail during 1994-95 (Jones and Langan, 1995c). Along the main channel of the Squamscott River, concentrations of fecal coliforms and *E. coli* increased dramatically going upstream from Chapmans Landing to the Exeter WWTF discharge pipe. Bacterial contaminants were present in relatively high concentrations in some of the fifteen small tributaries sampled along the Squamscott River, and analysis of salinities and bacterial contaminants suggested that the tributaries were affecting contaminant concentrations between Chapmans Landing and the upper reaches of the tidal river. However, there was no evidence for significant influence on water quality by any one tributary on the Squamscott River. Samples collected from ten sites in the freshwater Exeter River and tributaries showed higher concentrations in the downstream area near downtown Exeter. In a follow-up study, bacterial concentrations in the freshwater tributaries to the Exeter and Squamscott rivers were found to be elevated above state standards during dry and wet weather, with more severe contamination during wet weather (NHOSP, 1995a). The sites with higher concentrations in the lower portions of the Exeter River close to downtown Exeter were affected by stormwater runoff, and were suspected to be affected by septic systems and agricultural runoff (Becker and Radacsi, 1996).

An earlier study focused on the area from the Exeter River dam to Adams Point during 1989-90 (Jones, 1990). Prior to February, 1990, elevated bacterial concentrations in the Squamscott River were

Temporal trends for geometric means of fecal coliforms (colonies/100 ml) at three sites in the Great Bay Estuary at high tide.

FIGURE 2.5

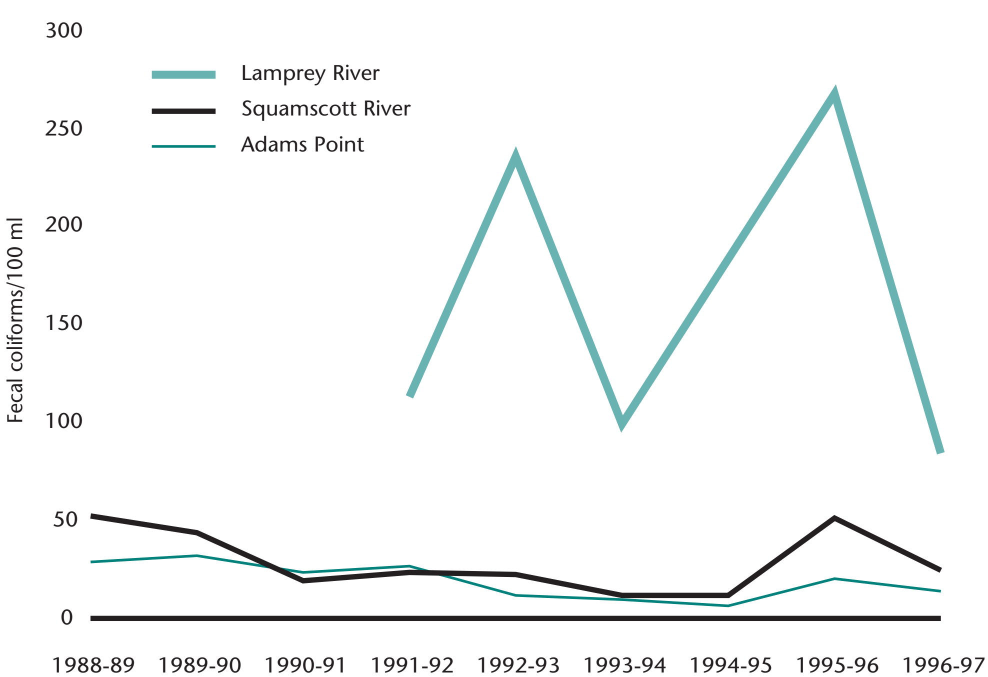


FIGURE 2.6

Geometric mean fecal coliforms (colonies/100 ml) in water collected during dry weather and storm events for three consecutive years in tributaries to the Great Bay Estuary: 1993-96, freshwater.

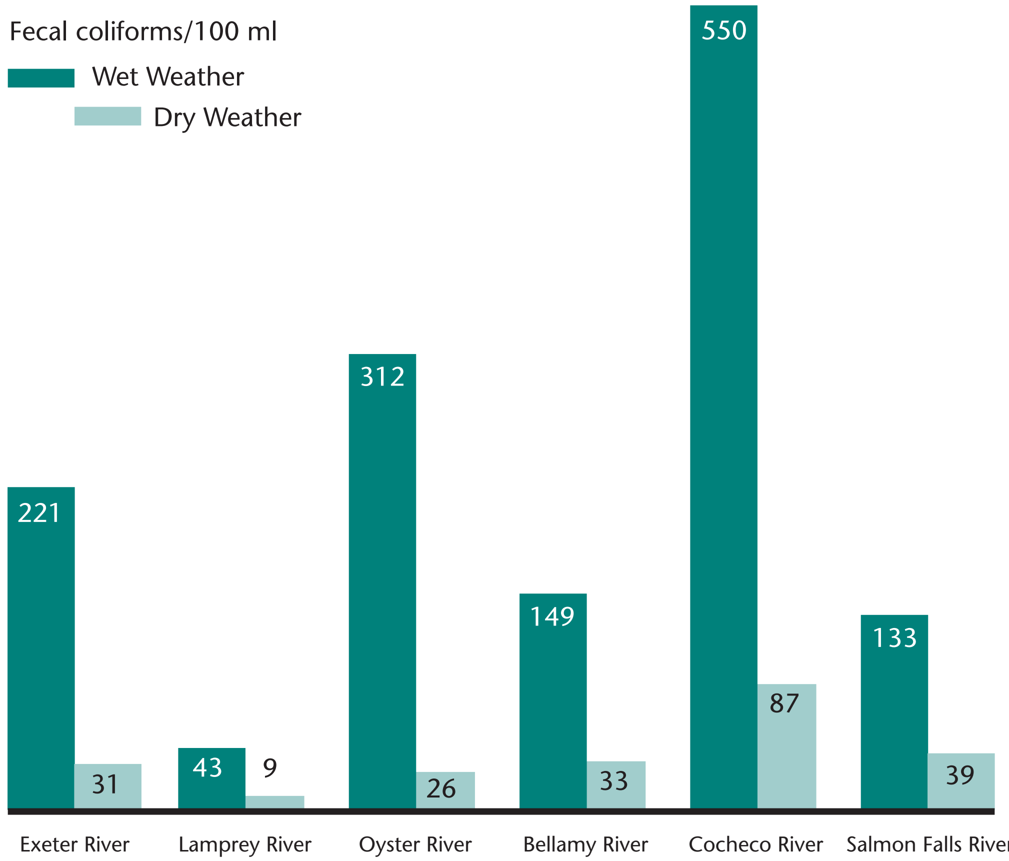
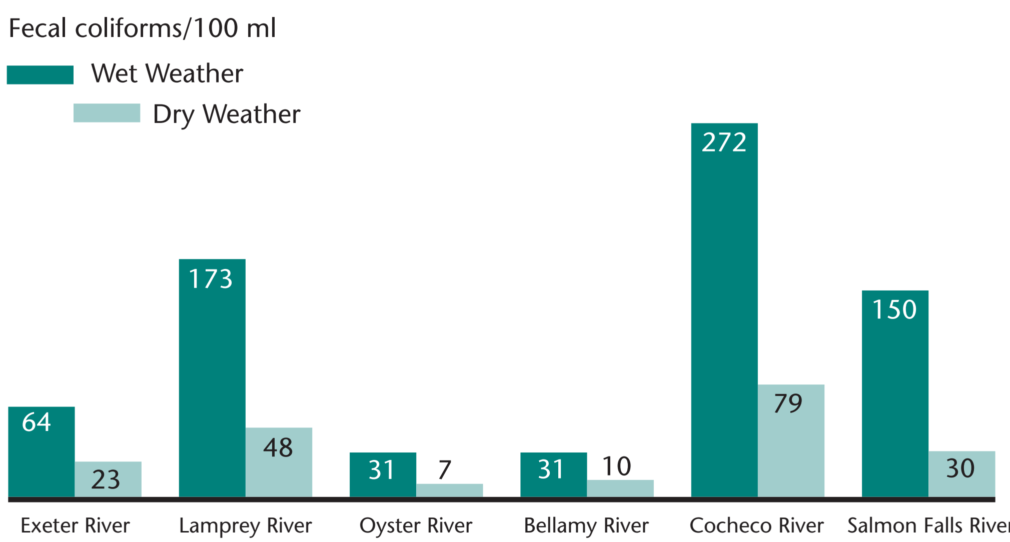


FIGURE 2.7

Geometric mean fecal coliforms (colonies/100 ml) in water collected during dry weather and storm events for three consecutive years in tributaries to the Great Bay Estuary: 1993-96, tidal water.



dominated by discharges from the Exeter WWTF. Water quality in the Squamscott River and Great Bay improved following the upgrading of the facility in early 1990. The concentrations of fecal coliforms, *E. coli* and enterococci discharged from the WWTF were high (105-106/100 ml) prior to the upgrade, and decreased to low levels (< 4/100 ml) thereafter. A comparison of indicators demonstrated the misleading nature of the total coliform assay. The organisms dominating a positive test value of 3000 total coliforms/100 ml in effluent collected after the upgrade when other indicator concentrations were nondetectable were identified as *Hafnia*, *Citrobacter* and *Aeromonas* sp., all common environmental species not associated with feces. These data were used as part of the justification by the state to discontinue use of total coliforms as an indicator of fecal contamination in surface waters.

Oyster and Bellamy Rivers and Lower Little Bay

This area extends from the freshwater portions of the two rivers through the tidal portions and into Little Bay from Fox Point to the General Sullivan Bridge (Figure 2.8). In the Oyster River, the DES and DHHS database results have been augmented by more detailed UNH studies (Jones and Langan, 1994c; 1993a; Margolin and Jones, 1990) and a recent study by NHCP (NHCP, 1996). NHDHHS data for 12 sampling stations in and around Great and Little bays were reviewed and interpreted as part of the 1995 sanitary survey (NHDHHS, 1995; Jones and Langan, 1995b). Fecal coliform concentrations were low enough to support an

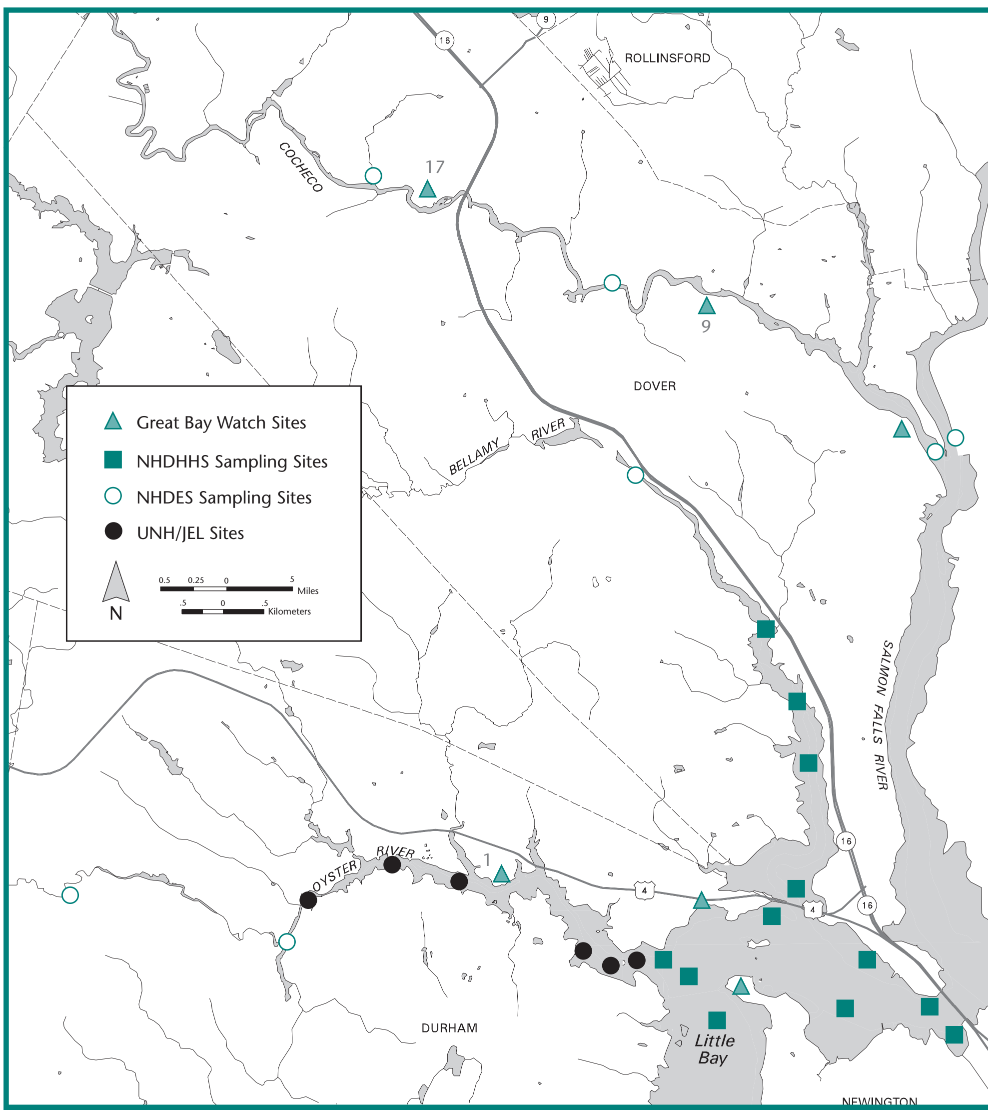


FIGURE 2.8

Oyster River, Bellamy River and Lower Little Bay water quality sampling region.

approved re-classification for the area in Little Bay that was monitored, which included two new sites during 1995-96 near Mathes Cove and Langley Island. Elevated concentrations near the mouth of the Oyster River only supported a restricted classification. Major rainfall events had significant negative effects on water quality and were noted as a potential condition for classification. Dye studies for the Durham WWTF and for the Great Bay Marina, conducted by USEPA in 1996 and 1997 (reports in preparation), will provide needed data to better define safety zones around these sites.

A new sanitary survey focused more intensive monitoring, including four new sites, in lower Little Bay (NHDHHS, 1998). Sanitary survey work was also performed in the Bellamy River and the analysis of fecal coliform data has been published (Jones, 1998a). The shoreline survey and fecal coliform concentrations at five of the six sites were consistent with an approved classification of much of lower Little Bay. Initially, only an area around Broad Cove was classified as approved, as other areas required additional samples. In June, 1998, as part of an amendment written to the original sanitary survey, most of the rest of lower Little Bay was re-classified as approved, except for an area from the mouth of the Oyster River east to Fox Point and areas around the two marinas.

Margolin and Jones (1990) found elevated concentrations of bacterial indicators in the Town Landing area of the Oyster River, especially following rainfall events. Geometric mean fecal coliform concentrations were >14/100 ml at six sites along the length of the river, except the WWTF outfall which had residual chlorine that disinfected the effluent and the river at the pipe. Poliovirus was also detected in 10 of 60 samples at six sites in the Oyster River, suggesting that sewage-borne viral pathogens could be present. There was no relationship between viral detection and concentrations of bacterial indicators.

The Oyster River Nonpoint Source Pollution Assessment project presented a comprehensive assessment of nonpoint source pollution in the Oyster River watershed, with emphasis on the tidal portion of the river and the tributaries that empty directly into the tidal river (Jones and Langan, 1993a). Fecal-borne bacteria levels were elevated in the watershed, and the levels in the tidal area were as high or higher than measurements made in other tidal rivers in the Great Bay Estuary. The geometric mean for fecal coliforms for all tidal sites was 37 FC/100 ml, which is consistent with a restricted or conditionally approved shellfish harvesting classification.

Fecal coliform and enterococci concentrations were highest in the Town Landing area, in Mill Pond and upstream in the tidal tributaries. Extensive sampling in the Beards and Johnson Creek watersheds showed elevated concentrations of bacteria throughout these watersheds. The bacterial contamination was dominated by nonpoint sources suspected to be on-site private sewage disposal systems (OSDs) and associated groundwater flow, urban and agricultural surface runoff, and other as yet undetermined sources. The evidence for these sources was based on elevated bacterial and nutrient contamination in some areas (Deer Meadow and Beards creeks) of the shoreline of the tidal river (suspected source: OSDs), areas within some tributaries where no direct source is apparent (suspected sources: groundwater flow, wildlife), consistent elevated responses to rainfall/runoff, and site-specific sampling around a farm where horses graze in and around a tributary. However, there is also some evidence to suggest that the Durham WWTF and some sewer lines are intermittent sources of significant contamination in water bodies that are crossed by sewer pipes.

The JEL study was continued for a second year, with more emphasis on the Johnson and Beards Creek watersheds (Jones and Langan, 1994c). Fecal coliforms, enterococci and *C. perfringens* concentrations were measured at fifteen sites along the tidal portion of the Oyster River. The highest concentrations were again detected in the upper reaches of the river near the Town Landing, with decreased fecal coliform and enterococci concentrations near the WWTF outfall caused by residual chlorine in the effluent. *C. perfringens* concentrations were highest near the WWTF outfall because their spores are resistant to chlorine disinfection. Elevated concentrations of bacterial indicators were again measured in the two watersheds, and a detailed study of salinity and fecal coliforms suggests that mixing of high concentrations in freshwater with cleaner salt water reduces bacterial concentrations in water beyond dilution effects. Expansion of sample sites into some branch brooks in the Johnson Creek watershed showed high concentrations around some housing developments that depend on septic systems, with one site contaminated by an identifiable residential septic system. In the more urban Beards Creek watershed, houses still on septic systems or leaky sewer lines were probably the sources of bacterial contamination. In fact, a small study at the mouth of Beards Creek gave clear evidence of contamination from a sewer line that crosses the mudflat. The latter and other identified sources of bacterial contaminants have been investigated by NHDES.

In a more recent study, data supported conclusions that the lower portion of the Oyster River watershed around downtown Durham is where most contamination occurs (NHCP, 1996). This study included sampling sites in the upper portions of the watershed and in the College and Pettee Brook areas that were not included in the JEL studies. Septic systems/leaky sewers and urban and agricultural runoff were probably the main sources of bacterial contamination. Sampling at most sites during storm events showed elevated bacterial concentrations, often exceeding 100 *E. coli*/100 ml, and sometimes exceeding 1000/100 ml for some sites.

Samples were collected at sites in the freshwater and tidal areas of the Bellamy and Oyster rivers as part of a three-year study to investigate the effects of storm events in tributaries to the Great Bay Estuary on water quality in the estuary (Jones and Langan, 1996a). The geometric mean concentrations of fecal coliform were relatively low during dry weather over the three year study at freshwater sites in both the Oyster (26/100 m) and the Bellamy (33/100 ml) rivers (Figure 2.6). The concentration in the tidal waters were low in both rivers (<11/100 ml) during dry weather (Figure 2.7). Concentrations increased significantly at all four sites, especially the freshwater sites, during storm events.

Salmon Falls, Cocheco, and (Upper) Piscataqua Rivers

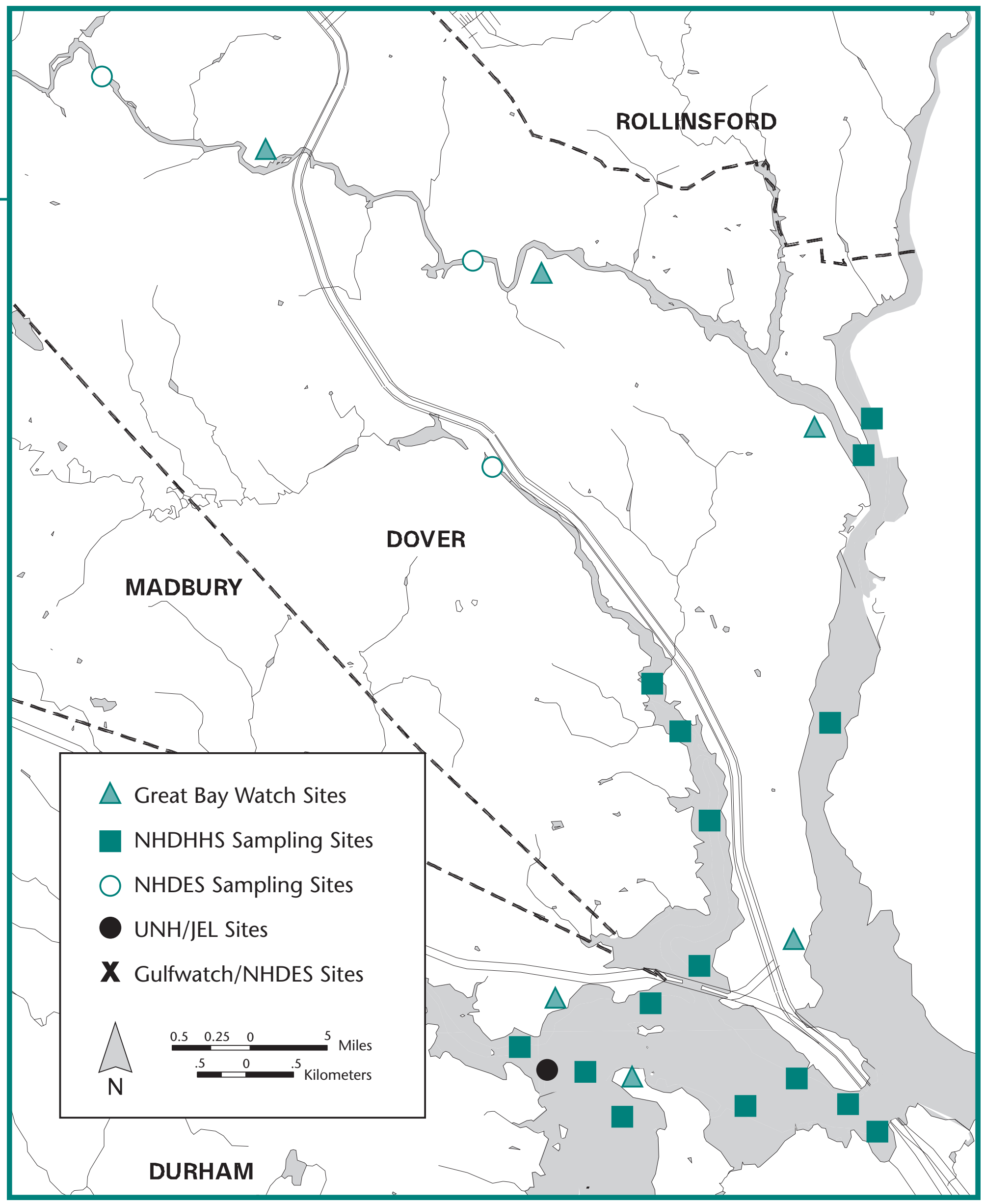
This area includes all estuarine and associated freshwater waters north of where Little Bay and the Piscataqua River meet near Dover Point (Figure 2.9). In the upper Piscataqua, Cocheco and Salmon Falls rivers, the DES and DHHS databases are augmented by some UNH studies, as well as State of Maine and Spinney Creek Shellfish Co. monitoring results (Mitnick and Valleau, 1996; Livingston, 1995). Sites in the freshwater and tidal areas of the Cocheco and Salmon Falls rivers were studied as part of the three-year investigation on storm events in tributaries to the Great Bay Estuary (Jones and Langan, 1996a). The geometric mean fecal coliform concentrations were elevated compared to other tributaries during dry weather over the three year study at freshwater sites in both the Cocheco (87 FC/100 ml) and the Salmon Falls (39 FC/100 ml) rivers (Figure 2.6). The concentration in the tidal waters were low in the Salmon Falls (30 FC/100 ml) and high in the Cocheco (79 FC/100 ml) during dry weather (Figure 2.7). Concentrations increased significantly (all >100 FC/100 ml) at all four sites, especially at the freshwater sites, during storm events. Some attenuation of bacterial concentrations apparently occurs between the upper and lower tidal portions of the Cocheco River, based on samples collected during 1997 (Reid et al., 1998). Even lower concentrations were measured downstream in the Piscataqua River. Lower bacterial concentrations were measured at a more upstream site in the Cocheco River. The high concentrations of bacteria in the downtown and downstream portions of the river suggest that urban areas of Dover are major sources of contaminants to this area of the Estuary, especially during storm events.

More recent studies have focused on contaminants in storm drains in downtown Dover and Exeter (Jones et al., 1999; Jones, 1998). All of the drains had detectable microbial contaminants during dry and wet weather. Levels of contaminants in street runoff were relatively low, suggesting that sources within the stormdrain system, probably illicit connections and leaking sewer pipes, were the major sources of the microbial contaminants. Contaminant concentrations in the Cocheco River were relatively lower during wet and dry weather compared to previous (Jones and Langan, 1996a) data.

Studies that focused on indigenous bacterial pathogens (i.e., vibrios) included assessments of fecal-borne bacteria (Jones et al., 1991a; O'Neill et al., 1990). Relatively high concentrations of fecal coliforms were detected in the Salmon Falls and Piscataqua rivers compared to Portsmouth Harbor during 1989-92. The general trend of higher concentrations of fecal-borne bacteria in tributaries was directly related to incidence of *Vibrio vulnificus* detection, but not for *Vibrio parahaemolyticus*.

FIGURE 2.9

Salmon Falls, Cocheco and upper Piscataqua rivers water quality sampling region.



Portsmouth and Little Harbors and Lower Piscataqua River

This area includes the Piscataqua River south of Dover Point, The Back Channel area and Portsmouth and Little harbors (Figure 2.10). In Portsmouth Harbor, Little Harbor, Back Channel and the lower Piscataqua River, routine NHDHHS and NHDES monitoring provides the most consistent databases, along with some limited UNH/JEL data. The data from the NHDHHS database have been summarized and interpreted relative to shellfish water classification standards in Jones and Langan (1996c), and more recent data are available (Appendix G). Sites in Little Harbor were generally in support of an approved classification, while fecal coliform concentrations were relatively high in Back Channel and tributary sites. Some areas in the Back Channel will probably be within a closed safety zone in the area around the Portsmouth WWTF effluent pipe.

A spatially intensive monitoring program to determine fecal contamination levels in water around Portsmouth Harbor, including some sites on the New Hampshire side, was conducted during 1992-93 (Jones, 1994). The sites were located along the main channel of the Piscataqua River. The geometric means for enterococci in the study area waters were generally consistent with safe recreational use criteria set by Maine and New

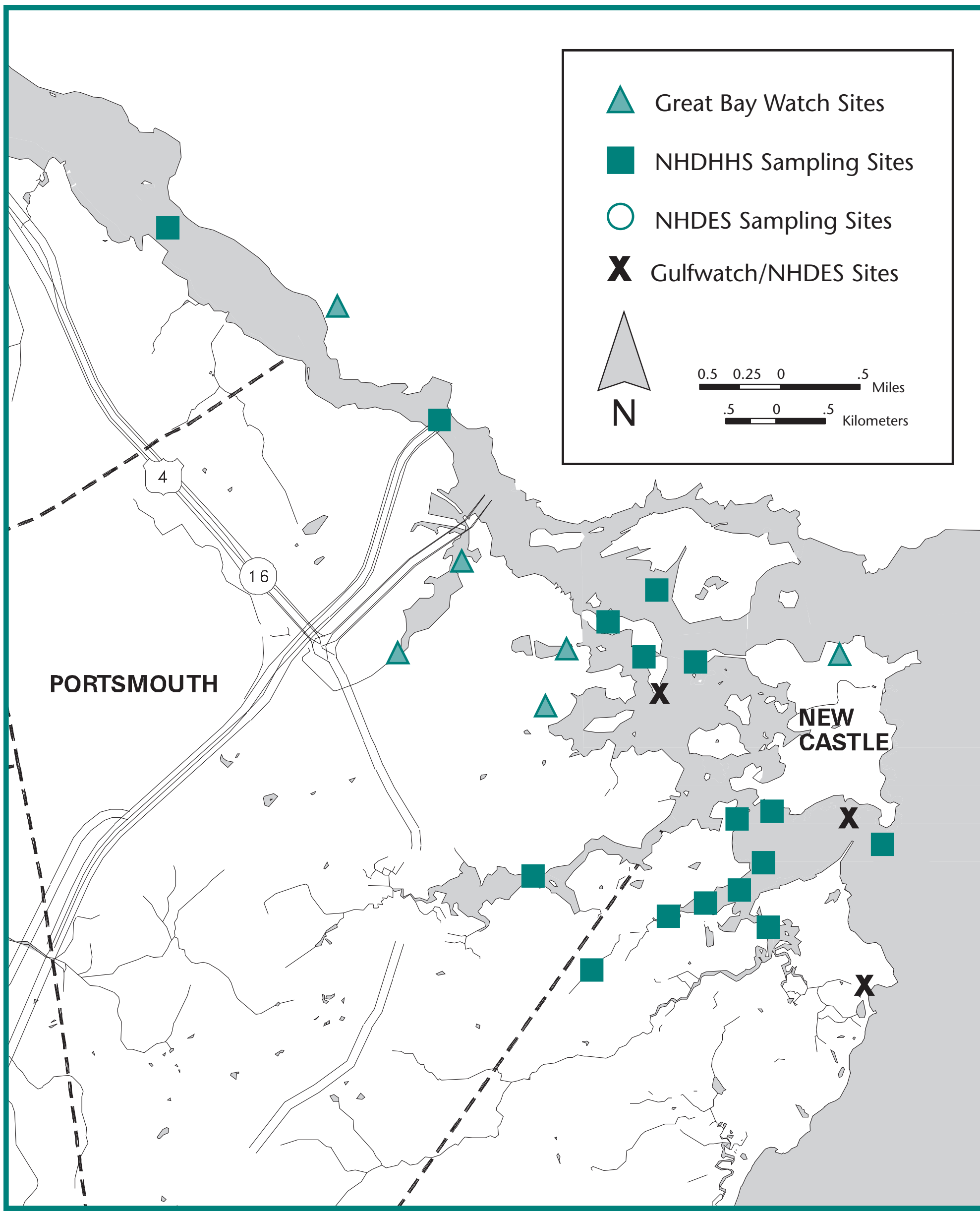


FIGURE 2.10

Portsmouth and Little Harbors and lower Piscataqua River water quality sampling region.

Hampshire (geometric mean <35/100 ml). The geometric means for fecal coliforms were all lower than the limit of 14 fecal coliforms/100 ml for approved shellfish-growing waters, but the frequency of samples greater than 43/100 ml was greater than 10% at the 6 stations. A long-term database (monthly for ten years) for samples from Ft. Constitution in New Castle has shown concentrations of fecal indicator bacteria to be consistently low at the mouth of the river (Dr. S. Jones, unpublished data). Four sites in North and South Mill ponds have been monitored for fecal coliforms saince 1997 by the Great Bay Coast Watch (Reid et al., 2000). Two one-year studies in North Mill Pond included fecal coliform measurements of the pond and storm drains (Jones, 2000; ANMP, 1998).

Rye Harbor and Coastline

This area includes the coastal areas from Little Harbor south to Hampton Harbor (Figure 2.11). In Rye Harbor and the coastline, existing data are mostly from NHDHHS and NHDES monitoring programs. Some of the data from the NHDHHS database have been summarized and interpreted relative to shellfish water classification standards in Jones and Langan (1996c), and more recent data are also available (Appendix G). NHDHHS data for some additional sites in tributaries are not presented, and NHDHHS data are summarized in Appendix G. The geometric mean concentrations of fecal coliforms at all four sites were <14/100 ml. However, the incidence of samples >43/100 ml was in excess of 10% in the

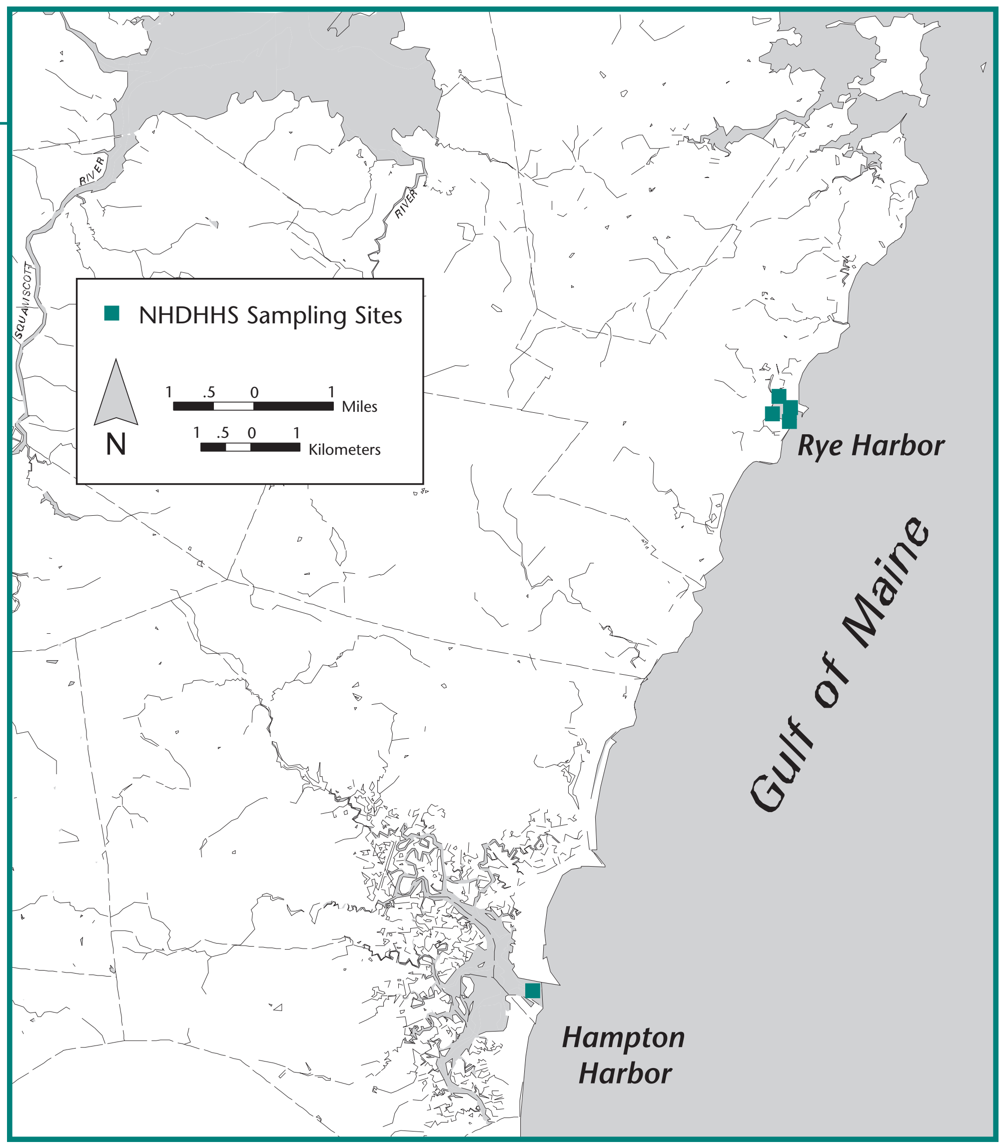


FIGURE 2.11

Coastal New Hampshire, from Little Harbor to the Massachusetts border, water quality sampling region.

last 30 samples at all but an inner harbor site, suggesting non-random contamination events are too frequent in the harbor to allow approved shellfish classification (NSSP, 1995). A boat pumpout facility has recently been put in at the NH Department of Resources and Economic Development (DRED) dock.

Hampton Harbor and Tributaries

This area includes all of the Hampton/Seabrook Estuary and tributaries (Figure 2.12). In Hampton Harbor, routine NHDHHS and NHDES monitoring, in cooperation with NHF&G, has provided long-term databases, while some recent more detailed UNH/JEL studies provide added information (Langan and Jones, 1995 a&b). The NHDHHS data for sites currently used for classifying shellfish waters in Hampton Harbor have been reviewed and interpreted (NHDHHS, 1994a), and more recent data are presented in Appendix G. The geometric mean fecal coliform concentrations for all ten sites were <14/100 ml. However, the incidence of concentrations >43/100 ml exceeds the standard 10% at some sites. Some of the sites with the more frequent incidence of high concentrations are near the mouth of Mill Creek on the west shore, suggesting that contamination from the creek may be influencing water quality in the area. Improved water quality in recent years has resulted in a recent upgrading of the shellfish harvest classification of the large Middle Ground clam flat in Seabrook from restricted to conditionally approved (NHDHHS, in prep.).

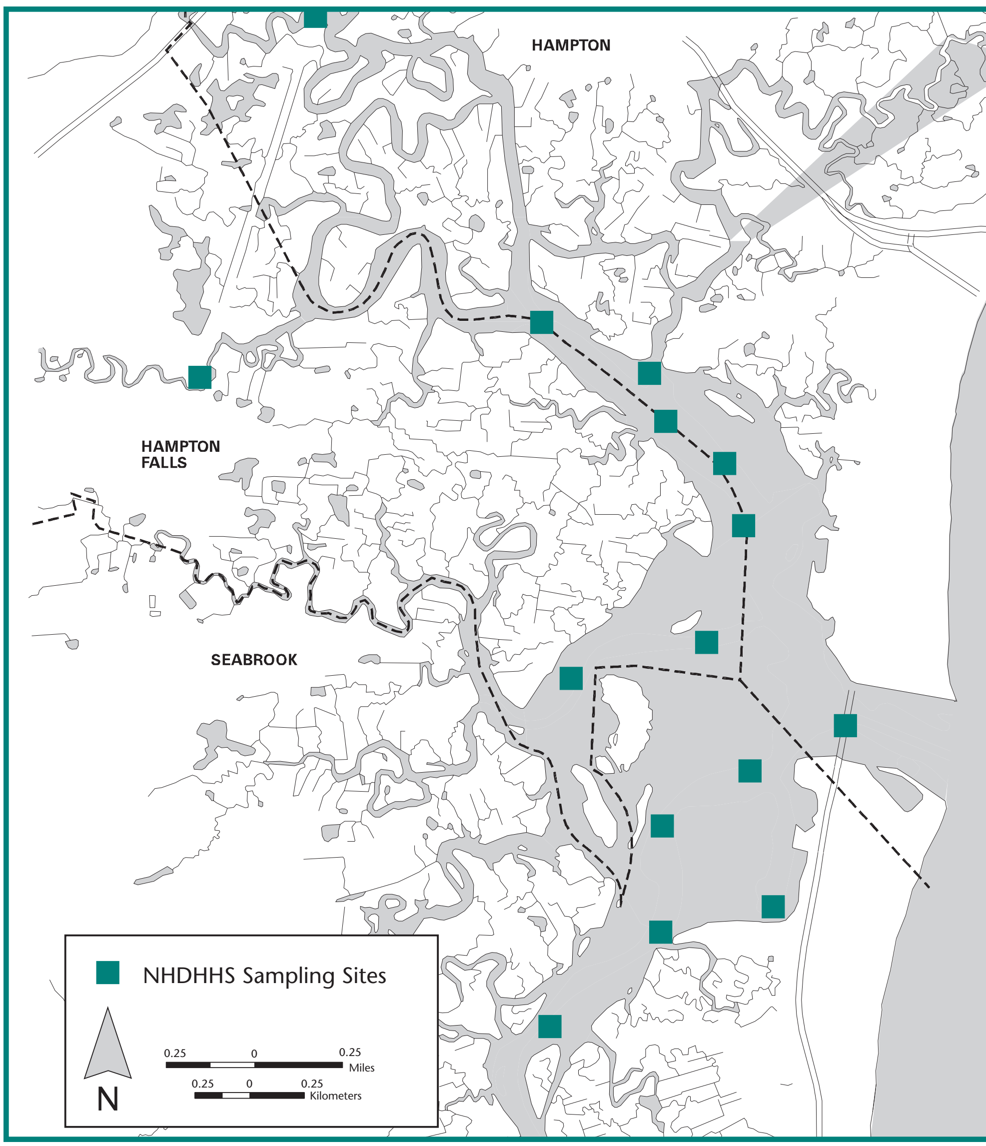


FIGURE 2.12

Hampton Harbor and tributaries water quality sampling region.

Water quality survey on Cocheco River

A two-year study on septic systems in Seabrook included some surface water monitoring, with emphasis on tributaries that border residential areas (Jones et al., 1995; 1996). Samples were collected from 16 sites at low tide in Mill Creek, Farm Brook, some tidal creeks and the harbor. Water from Mill Creek had the highest levels of indicator bacteria (<200 FC/100 ml) during sampling in 1995 and 1996. Concentrations of bacteria detected at all upstream tributary sites were elevated compared to harbor sites. Lower concentrations in the harbor were probably the result of dilution and die-off in the more saline waters, which represents less favorable conditions for bacterial survival. Seven sites, mostly in tributaries, did not meet the New Hampshire swimming water standard of 35 enterococci/100 ml. Based only on the study data, only one site had a mean fecal coliform concentration <14/100 ml. There was no clear relationship between groundwater contamination and surface water quality at any site, although the elevated concentrations of bacteria in streams near high density residential areas suggests septic systems are a likely source of contamination. During 1996-97 when septic systems were being disconnected and sewage was diverted to the new treatment facility, measurements of contaminants in the surface waters of the harbor and tributaries showed little change from previous years (Jones, 1997).

Clearly, there are sources of bacterial contaminants that persist in all areas of coastal New Hampshire and limit uses of estuarine and coastal waters. The concern is the protection of public health in areas that will only experience increased human use in the future. Continued efforts to identify and either eliminate or effectively manage the impacts of fecal contamination sources is an important, on-going issue in coastal New Hampshire. As the next section suggests, water quality in general has improved over the last ten years, but the widespread nature of the problem suggests that much remains unknown about the issue.

2.2.1.2 Temporal Trends

There appear to be some general temporal trends that have occurred in many areas of the Seacoast. Fecal-borne bacterial contaminant concentrations have decreased in all coastal waters since the early 1990s as a result of the extensive improvements to wastewater treatment facilities. Bacterial contaminants are also generally present at higher concentrations at low tide compared to high tide, mostly as a function of mixing of more contaminated freshwater with cleaner tidal water. Bacterial concentrations are often elevated during autumn and winter compared to other seasons in some areas. This observation is probably related both to the amount of runoff associated with rainfall events as a function of seasonal differences in evapotranspiration and infiltration, and to the enhanced survival of bacterial contaminants with colder water temperatures (Jones et al., 1997). The most severe incidences of elevated contamination occur in temporally less predictable conditions, i.e., following rainfall/runoff events and upsets in treatment processes at WWTFs. In addition, >100 year storms such as the one that occurred in October, 1996, tax the capacities of most WWTFs because of infiltration into the sewer systems and overloading of treatment plants. Some areas are more prone to contamination incidences because of proximity to WWTFs, especially those that may lack effective control measures for stormwater runoff and have less capacity for effective wastewater treatment during storm events.

Long-term trends for total coliform concentrations (per 100 ml) in water samples collected from six tributaries to the Great Bay Estuary, 1960, 1975, and 1996. **TABLE 2.3**

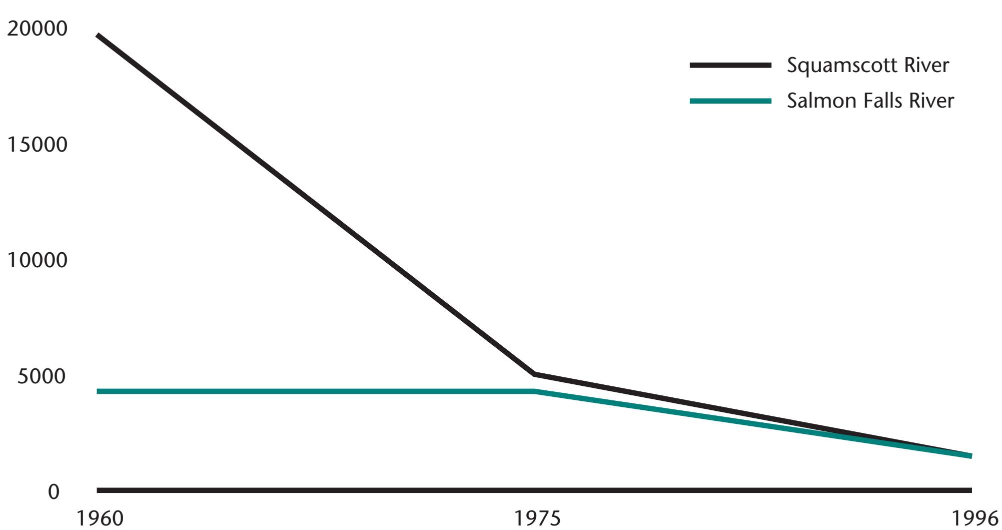
FRESHWATER SITES AT TIDAL DAMS

YEAR	Exeter R. 9-EXT	Lamprey R. 5-LMP	Oyster R. 5-OYS	Bellamy R. 5-BLM	Cocheco R. 7-CCH	Salmon Falls R. 5-SFR
1960	19700	524	656	—	16540	4266
1975	5044	1088	3742	4786	133690	4266
1996*	1490	350	1310	1345	1530	1475

*1996 data transformed by multiplying fecal coliform concentrations by 5.

Certain sites in coastal New Hampshire have been sampled for decades and the results can be used for determining temporal trends. Data from three reports (Jones and Langan, 1996a; NHWSPCC, 1975; NHWPC, 1960) are summarized in Table 2.3 to illustrate the dramatic improvements in water quality since 1960. Because the two earlier reports used total coliforms and the third used fecal coliforms, it was assumed that total coliform concentrations were equivalent to five times the fecal coliform concentrations, and the 1996 data were converted to total coliform equivalent data. This conversion is based on the relationship between total and fecal coliform standards for classifying shellfish growing waters (NSSP, 1995). The data show decreases in total coliform concentrations in all six rivers from 1960 to 1996. The decrease was most dramatic in the Cocheco River, which has remained the most contaminated tributary since 1944, but which showed a nearly 100-fold decrease from 1975 to 1996. The higher concentrations in 1975 compared to 1960 may reflect increased loading of wastewater treatment facilities due to the nearly doubling (158,800 to 275,800) of populations in Rockingham and Strafford counties from 1960 to 1980 (NHOSP, 1997a). There was also a dramatic, steady decrease in the Exeter/Squamscott River and a less extensive decrease in the Salmon Falls River (Figure 2.13). The following section summarizes in more detail existing information on the temporal trends of bacterial contamination in the different estuarine and coastal areas of New Hampshire. Where possible, discernable temporal trends are related to

Total coliforms (colonies/100 ml) in the Exeter/Squamscott and Salmon Falls rivers: 1960-1996. **FIGURE 2.13**

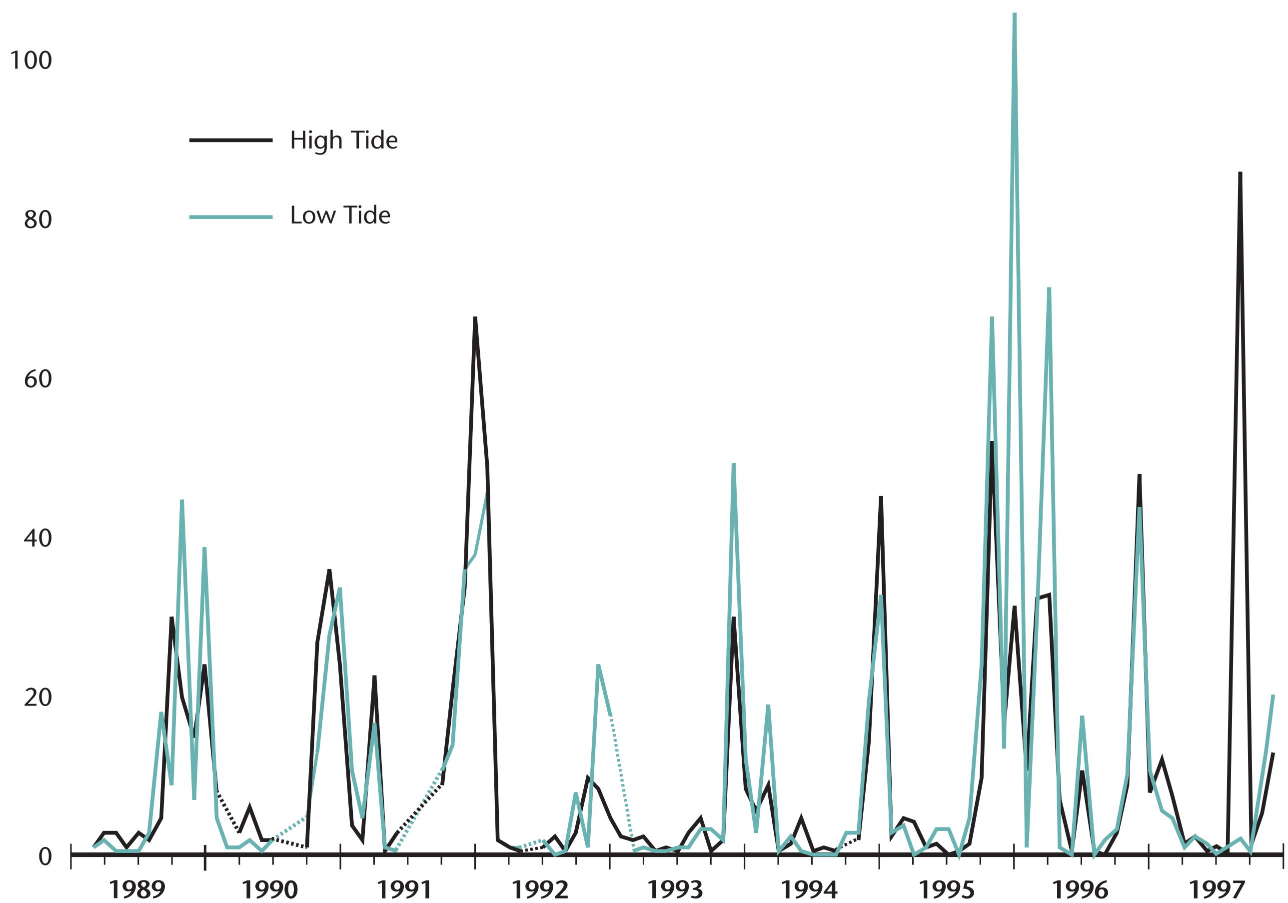


management efforts to reduce pollution.

The overall trend over the nine year period of GBNERR monitoring (Langan and Jones, 1997) has been a general decrease in bacterial contaminants at all sites (Figures 2.4 and 2.5), although concentrations of all indicators were higher during 1995-96 than during previous years. The three-year study of tributaries to Great Bay Estuary also showed some bacterial contaminants were present at significantly higher concentrations during 1995-96 compared to the previous two years in the Lamprey and Squamscott rivers (Jones and Langan, 1996a). The long-term decrease in bacterial concentrations was most dramatic in the Squamscott River, especially after 1990 when the Exeter WWTF was upgraded. Trends for fecal contaminants were less dramatic at other sites like Adams Point, where concentrations have been relatively low (<33 FC/100 ml) since 1988. It also appears that reducing concentrations much below the standard 14 FC/100 ml may be difficult when other areas continue to have higher concentrations. Seasonal trends show contaminants tend to be present in higher concentrations during late autumn and winter, as illustrated in Figure 2.14 for enterococci at Adams Point from 1989-97, which is consistent with runoff conditions and bacterial survival patterns (Jones et al., 1997). As previously mentioned, contamination trends at the Lamprey River do not follow typical patterns, as fecal coliforms are typically highest during the summer, instead of autumn/winter.

Various studies in the Oyster River were conducted from 1992-1997 (Jones and Langan, 1996a; 1994c; 1993a; Reid et al., 1998). The 1992-93 seasonal trends for enterococci showed a clear trend of elevated concentrations in summer, while fecal coliform concentrations

FIGURE 2.14 *Monthly concentrations of enterococci (colonies/100 ml) at high and low tides at Adams Point: 1989-1997.*



exhibited a mixture of trends at all sites (Jones and Langan, 1993a). The next year, seasonal trends for enterococci and fecal coliforms were mixed, while *C. perfringens* showed a clear trend of elevated concentrations during springtime for almost all sites (Jones and Langan, 1994c). In the Johnson Creek watershed, fecal coliform and enterococci concentrations were uniformly at much higher concentrations during summer and, to a lesser extent, autumn, compared to winter and spring. This may be the result of increased regrowth at higher temperatures and reduced flow during warm months. Rainfall events >0.25"/24 h caused elevated concentrations of enterococci at most sites and higher fecal coliforms at sites near the Town Landing. There has been an overall decrease in fecal coliform concentrations near the mouth of Bunker Creek from 1992-97 (Reid et al., 1998). At Mill Pond, fecal coliform and enterococci concentrations were decreasing from 1993 to 1996 during both dry and wet weather (Jones and Langan, 1996a). In the Bellamy River, fecal coliform and enterococci concentrations increased from 1993 to 1996 during both dry and wet weather.

In downtown Dover above the tidal dam, fecal coliform and enterococci concentrations exhibited mixed trends from 1993 to 1996 during both dry and wet weather (Jones and Langan, 1996a). In the tidal portion of the Cocheco River, fecal coliform and enterococci concentrations increased from 1993 to 1996 during both dry and wet weather. The trends for both enterococci and fecal coliforms were mixed for dry and wet weather at the freshwater and tidal sites in the Salmon Falls River.

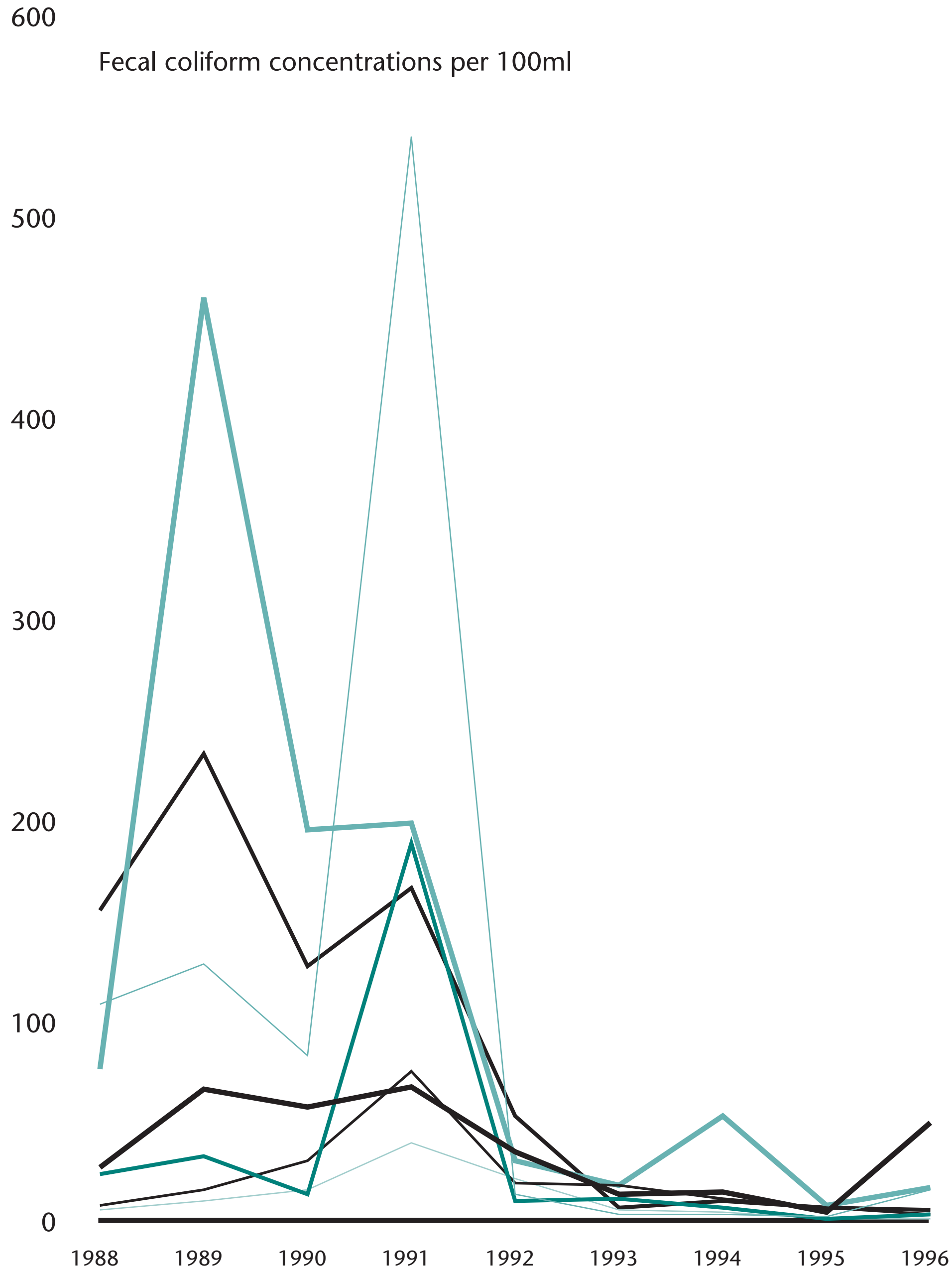
Temporal trends for fecal coliforms showed an overall decrease in concentrations since 1988, especially after 1991, in Portsmouth Harbor, Little Harbor, the Back Channel and the lower Piscataqua River (Figure 2.15). The striking decrease after 1991 was coincident with the construction of advanced wastewater treatment in Portsmouth. Continued detection of fecal coliforms at concentrations >14/100 ml are the result of lingering nonpoint sources and possibly the two CSOs remaining in Portsmouth. The contribution of the CSOs to contaminant loading is not known, although the CSOs discharge a combination of untreated sewage and stormwater during some storm events (NHDES, 1996a).

In Rye Harbor, concentrations of fecal coliforms have decreased at all sites since 1985, especially at the harbor mouth (see Appendix G). Lower concentrations after 1991 could have been the result of connection of some Rye residences to the Hampton WWTF.

The temporal trends for annual geometric mean fecal coliform concentrations in Hampton/Seabrook Harbor showed an overall decrease for all sites from 1988 to 1996. The lowest concentrations for 8 of the 10 sites occurred in 1995. Further improvements in water quality are expected to occur following the completion of connections of all present septic system sites in Seabrook to the new town sewer system. Improvements in the sanitary quality of the Harbor water was not yet apparent in mid-1997 after many of the areas adjacent to tidal waters had been connected (Jones, 1997).

The overall improvement in water quality relative to bacteriological measurements is a reflection of the significant resources expended to improve wastewater treatment facilities in coastal New Hampshire. Population growth continues at a slower pace relative to previous decades. The estimated increase in population in Strafford and Rockingham counties from 1990 to 1996 was 350,000 to 367,900, only a 5% increase (NHOSP, 1997b). Nevertheless, increases in human population, development, impervious surfaces with associated stormwater runoff, and wastewater treatment demands will continue to change the ability of watersheds to handle the additional pollution. A better understanding of the watershed factors that affect transport and fate of microbial contaminants would help frame effective strategies for eliminating or managing pollution sources and transport pathways for these contaminants to estuarine waters.

FIGURE 2.15 *Fecal coliform concentrations at seven sites in Little Harbor, Back Channel and Portsmouth harbor: 1988-1996.*



2.2.2 SOURCES OF FECAL-BORNE BACTERIA

By definition, fecal-borne bacteria are from the small intestines of mammals, and their presence is indicative of the presence of sewage and other fecal material. However, the bacterial indicators cited in this report that are used to assess sewage contamination; total and fecal coliforms, enterococci, *E. coli* and *C. perfringens*, may be found in other animals and are all capable of existing outside of the small intestine and may be found to occur naturally in the environment. Thus, caution is required when interpreting the fecal indicator data in efforts to identify sources of pollution. Ongoing studies by UNH/JEL and NHDES are focused on developing methods (Parveen et al., 1999) to identify specific sources of fecal indicator bacteria.

Prior to the efforts in the late 1980s and early 1990s by New Hampshire to upgrade all WWTFs in the Seacoast, point sources were the major source of bacterial contaminants in the Great Bay Estuary and coast. More recently, the masking effects of point source pollution have been drastically reduced to occasional malfunctions or storm event overloading at WWTFs, and nonpoint source pollution is now the major source of chronic contamination.

A summary of the recent status of sources of bacterial contaminants in shellfish waters was compiled by NHDES (NHDES, 1995). It lists WWTFs, CSOs, and urban stormwater as the major sources of bacteria, and unidentified nonpoint sources as important in some areas. In the following section, the existing information on these and other sources will be described.

2.2.2.1 Storm-related Runoff

The most common source of bacterial contamination in New Hampshire is runoff resulting from rainfall/snowmelt events in urban and urbanizing areas. This conclusion is based on the elevated concentrations of bacteria detected in all areas following rainfall events and the proximity of urbanized areas to tidal water sampling sites, as reported in almost every recent study. Some reference to stormwater effects in the different areas have already been cited.

The best illustrations of the impact of storm events on surface water quality are some recent projects conducted by JEL. The first is a three-year study on the effects of storm events on water quality in the tributaries of the Great Bay Estuary, as summarized in Jones and Langan (1996a). Statistical analysis of the cumulative 3-year data showed significantly higher bacterial concentrations following storm events at every freshwater and estuarine site (Figure 2.6 and 2.7). The freshwater sampling sites were all located at the tidal dams, all of which are located within urbanized areas of the nearby municipalities of S. Berwick, ME and Dover, Durham, Newmarket and Exeter, NH. More detailed studies of the watersheds around the Exeter (Jones and Langan, 1995c; NHOSP, 1995a) and the Oyster (NHCP, 1996; Jones and Langan, 1993a; 1994c) rivers have confirmed that urban runoff is an obvious source of contamination in these areas. This issue is presently being addressed by support from the NHEP and other ongoing projects. Some municipalities have inventories of stormwater outfalls. Those that have inventories include Greenland and parts of Dover, Rochester and Seabrook. However, the quantity and quality of the information varies, making it difficult to formulate a clear picture of the magnitude of stormwater outfalls as potential pollution sources.

A better understanding of contaminants in stormwater runoff has been recently emerging. NHDES (1997) found significant dry weather contamination in stormwater pipes draining into the Cocheco and Squamscott rivers. A follow-up study included wet and dry weather sampling in the Bellamy and Cocheco rivers (Landry, 1997). Significant contamination was observed in the Cocheco storm drains during dry weather and the Bellamy drains in wet weather. More comprehensive studies by Jones (1998) and Jones et al. (1999) focused on the worst of the drains on the Cocheco River and showed contaminants flowed from the drains continuously during dry and wet weather, in some cases at high concentrations.

Other recent studies on stormwater contamination have been designed to assess the effectiveness of stormwater control measures. Jones and Langan (1996b) focused on ten different stormwater control systems in the NH Seacoast region during 1995-96, including swales, retention ponds, a pond with staggered dikes and an infiltration chamber. First flush (during the first 0.25 inches of rainfall) samples were analyzed for a variety of contaminants, including bacterial indicators. Results showed that wet ponds were more consistently effective at treating diverse contaminants than swales. During summer, bacterial concentrations increased both in influent and effluent water, and all systems were less effective at removal. The results suggest that bacteria may re-grow in the moist, nutrient-rich control systems during dry periods that occur between storms. Elevated concentrations are then discharged with new storm events. This raises the issue of the public health significance of stormwater runoff. It also suggests that some system designs may not be effective in treating bacterial contaminants. A follow-up study (Jones, 1998c) of five systems during dry weather showed evidence of some growth occurring during summertime in some systems and suggested certain conditions may be conducive to growth.

The 1996 New Hampshire Water Quality Report to Congress 305(b) (NHDES, 1996b) reported that 17.3 square miles of coastal estuaries are not fully supporting uses because of pathogen indicators, and that the source of bacteria is unknown. It states that stormwater runoff is a well-documented source of bacteria and nutrients, citing numerous studies (Jones and Langan, 1996a; 1996b; NHCP, 1996; Swift et al., 1996). Stormwater was also cited as a significant source in coastal New Hampshire in another DES report (NHDES, 1995). The 305(b) report also pointed out that rainfall is a condition for closure of Hampton Harbor because of runoff-associated bacteria, as reported in the sanitary survey (NHDHHS, 1994b).

Other studies in New Hampshire have shown degradation of surface water quality from rainfall runoff. The runoff water from seven storm events in two developed areas in Concord had fecal coliform concentrations ranging from 23 to 240,000/100 ml (NHWSPCC, 1979). A more recent study (Comstock, 1997) found *E. coli* concentrations in stormwater runoff consistently exceeded state water quality criteria at both an urban and a residential site. Water quality in Great Bay was reported to be degraded during periods of high rainfall and runoff (NHDHHS, 1992). Several street drainage systems in Hampton and drainage ditches in Seabrook, some of which contained fecal contaminants, were found to drain directly into the marsh and tidal waters of Hampton Harbor (NHDHHS, 1994). NHDES (1997) also reported stormdrain catch basins with high *E. coli* concentrations in Hampton.

The most intensive study on stormwater was conducted by the NH Water Supply and Pollution Control Commission (NHWSPCC) in 1983 as part of the EPA Nationwide Urban Runoff Program (Oakland, 1983). The impacts and methods for control of stormwater were studied in tidal and freshwater portions of the Oyster River watershed in Durham, NH. Water quality in the watershed declined significantly following storm events, especially for total and fecal coliforms. Because Durham maintains a separate stormwater and sanitary sewer system, sources of contaminants during storms were suspected to be from animal feces. Sources for dry weather contamination were not identified. Studies on stormwater runoff control measures showed favorable effects on bacterial contamination with parking lot vacuum cleaning and a river-run impoundment (Mill Pond), but not with a grassed swale. The grassed swale showed significant removal of inorganic nitrogen, but orthophosphate and bacteria concentrations increased. The river-run impoundment, in contrast, showed significant removal of mass loads for bacteria and inorganic nitrogen, with a non-significant increase in orthophosphate, with length of detention time a positive factor.

The major Best Management Practices (BMPs) used to control urban runoff in New Hampshire in 1989 were treatment swales and sedimentation basins (NHDES, 1989a). The report suggested that these control measures are effective for trapping sediments, controlling erosion and removing some heavy metals. However, the report recognized these systems as being ineffective at treating nutrients, bacteria, oil and suspended solids. New rules for stormwater control measures for large developments have been adopted, and a new manual describing acceptable control systems has been published (NHDES, 1996). The effectiveness of each type of system for treating a range of different contaminants is presented, along with advantages, disadvantages and design criteria.

Stormwater runoff is considered to be a serious nonpoint source pollution concern by 68% of polled residents of the Oyster River watershed (Hanratty et al., 1996). Even though 87% said that problem storm drains should be upgraded, they were largely unwilling to pay for corrective actions. NHDES estimated that rehabilitation of coastal collection systems and treatment of stormwater would cost $100-200 million (NHDES, 1995), and that the chances of successful treatment of bacterial contaminants is slim. For ongoing work in the Seacoast, NHDES considers this issue a significant problem, and it is a major focus of the latest NHDES Coastal Basin Nonpoint Source Pollution Assessment and Abatement Plan (NHDES, 1996a). Present efforts by NHDES and UNH/JEL are focused on investigating stormwater systems during dry and wet weather, and following up on problems in tributaries to coastal rivers identified in previous JEL, NHOSP, NHDHHS and NHDES studies.

Unlike previous studies that often conclude that animal feces is the major source of microbial contaminants in stormwater runoff from urban areas, the major source of contaminants in New Hampshire coastal urban runoff appears to be direct sewage contamination from leaking pipes and illicit connections. Thus, even though there may be separate sewage and storm drain systems, their age, design and close proximity below the surface appear to be conducive to cross contamination.

2.2.2.2 Wastewater Treatment Facilities and Combined Sewer Overflows

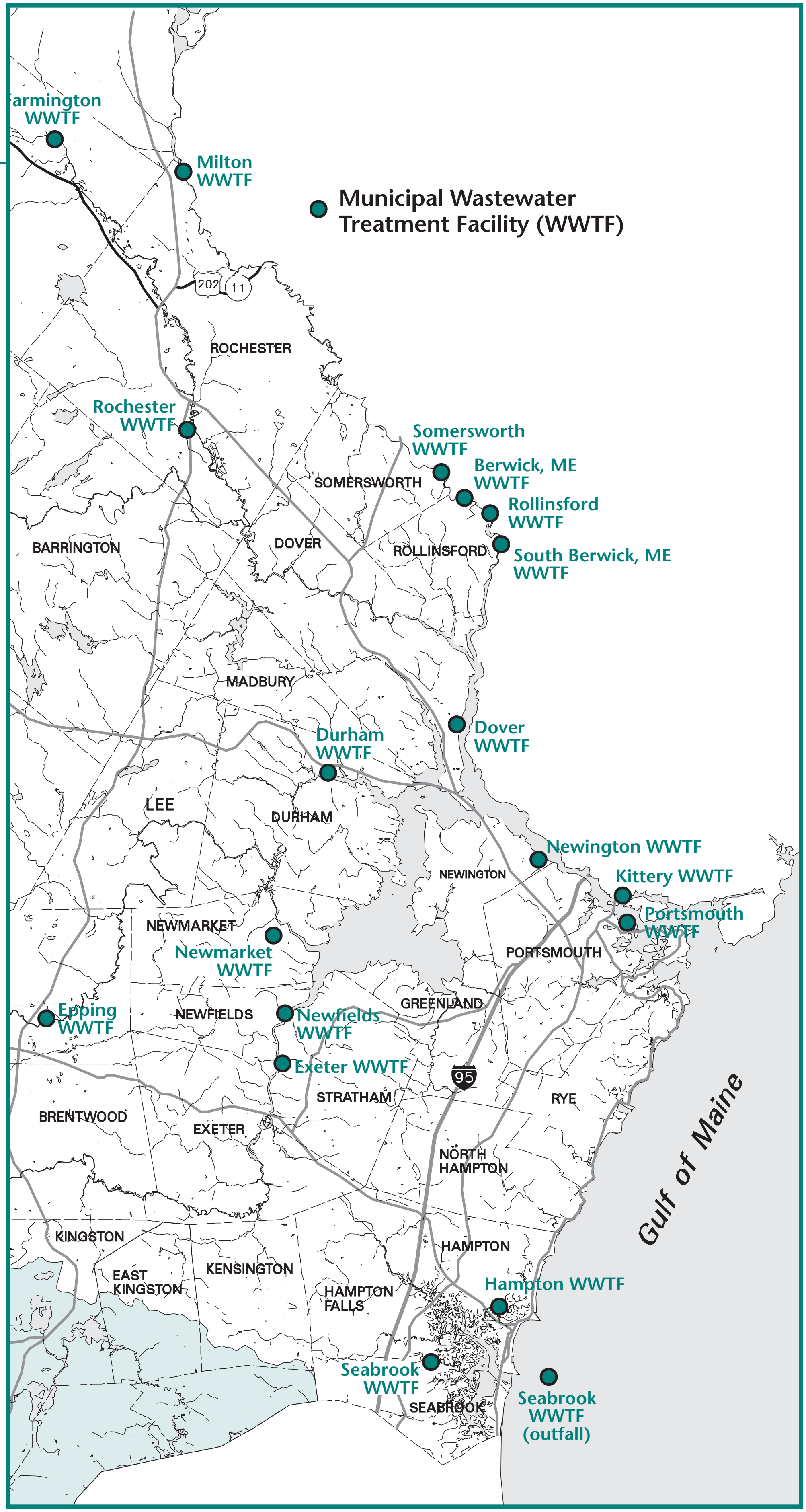
WWTFs are, ideally, capable of reducing microbial contaminant concentrations to meet required criteria in wastewater 100% of the time. However, this does not occur in practice. Changes in waste stream characteristics that modify treatment efficiency, equipment problems, operational changes, human error and acts of God (hurricanes, lightning, storms) all influence the effectiveness of WWTFs. The WWTFs in New Hampshire and their effluent flow ranges are presented in Figure 2.16. NHDES records the number of upsets that facilities report, although documented impacts of upsets in treatment processes on surface water quality are rare (Jones and Langan, 1993a; 1994c). Reporting of upsets has increased in recent years resulting in better characterization of the problem (NHDES, unpublished data). WWTFs report upsets to NHDHHS so shellfish areas can be closed. All coastal WWTFs have a limit of 70 total coliforms/100 ml at discharge pipes, they are required to conduct daily testing and chlorine residuals are required to be low/non-toxic. A few WWTFs still have problems meeting the total coliform discharge limit, and modifications to disinfection systems are being planned for most of these systems.

Some coastal WWTFs and sewer systems have limited capacities for handling stormwater during major storm events. Stormwater can overburden facilities and require bypassing of pump stations. Under these conditions, inadequately treated wastewater is discharged to tidal waters and significant loading of bacteria can occur. This happens several times each year and shellfish beds downstream from the affected facilities have been closed. The ‘100 year’ storm of October, 1996 caused bypasses in all but a few coastal WWTFs. Other stormwater related problems include infiltration of stormwater and high groundwater into sewer pipes. This may result in leakage of pipes. It is suspected to be a problem in all urban areas, and has been documented in Durham (Jones and Langan, 1994c). The problems and the extensive documentation of high levels of contamination in tidal waters following major storm events are the basis for closing the whole coastal area to shellfishing until water quality returns to acceptable levels and shellfish have depurated contaminants. The state has made many improvements in WWTFs throughout the coastal area (Table 2.4), and these efforts continue (NHDES, 1996d).

The two remaining CSOs in Portsmouth are significant sources of bacteria that impact the water quality of

FIGURE 2.16

Municipal wastewater treatment facilities.



Little and Portsmouth harbors. Portsmouth has eliminated eight of ten CSOs, but two remain in South Mill Pond. A concern for the Little Harbor area is that contaminants flushed into South Mill Pond from the CSOs could flow through the Back Channel area into Little Harbor (NHDES, 1995). Elimination of the remaining CSOs would cost an estimated $10 million, as estimated by the city's CSO Facility Plan. Because of the high costs associated with elimination of the CSOs, the City of Portsmouth has filed for a Use Attainability (UAA) Study to reclassify the receiving waters, i.e., South Mill Pond. If they are successful in proving that the costs are essentially prohibitive, then they would not be required to attain the limit of 70 total coliforms per 100 ml in South Mill Pond. In such a case, careful attention to the potential for storm-related contamination to affect any opened shellfish beds in Little Harbor would be necessary. It would also be difficult to open the extensive mudflats in the Back Channel area.

Point source pollution control program activities from 1988-1996: WWTFs and CSOs. **TABLE 2.4**

City	Wastewater flow (mgd) design	ave.*	max.*	Control measure	Date completed	Cost
Dover	4.4			new 2° treatment facility	1991	$24,300,000
Strafford Co. Facility				cease discharge to Cocheco R.	1992	
Durham	2.5	1.0	4.5	upgrade from 1° to 2° treatment	1981	
				equipment upgrades	1992-93	
				dechlorination	1995	
Exeter	3.0	1.6	6.2	lagoon system built; dechlorination	1990	$5,900,000
				all but one CSO disconnected	1992	$3,400,000
Farmington	0.4			secondary clarifier	1994-95	
Hampton	3.5			sewer project and dechlorination	1993	$4,400,000
Newfields	0.1	0.04	0.2	construction of facility	1983	
Newmarket	0.9	0.6	2.5	upgrade from 1° to 2° treatment	1986	$1,900,000
				dechlorination/dewatering system	1993	
Newington	0.3			upgrade disinfection system	1995	~$350,000
Portsmouth	7.0			new advanced 1° treatment & dechlorination	1992	$15,000,000
				eliminate 10 CSOs	1991	$5,800,000
Rochester	3.9			currently designing new advanced treatment		
Rye				sewers connected to Hampton POTW	1991	$2,400,000
Wallis Sands St. Pk.				UV disinfection; refurbish sand filter	1993	
Seabrook				construction of wastewater treatment facility	1995	
Somersworth	2.4			various improvements; P reduction study		
Star Island				construction of seasonal 2° treatment plant	1994-95	

* in 1994

Ongoing work is focusing on a hydraulics study of the CSOs around South Mill Pond, identification and elimination of illicit connections and dye studies of the WWTF outfall pipe. A safety zone around the outfall pipe will probably extend into the nearby Back Channel.

One CSO remains in Exeter. The CSO is a source of bacteria during storm events when the capacity of the main pump station is exceeded. Under those conditions, sewage can overflow into Clemson Pond, which acts as an emergency holding pond. However, the water that drains from the pond to the Squamscott River is often contaminated (NHOSP, 1995; Jones, 1990).The problem is currently under investigation. Exeter passed a warrant article in 1999 to allocate $1.7 million to address the CSO problem.

As previously stated, the system of wastewater treatment facility pipes that transport sewage from sources to the treatment plant are a potentially significant source. In several coastal New Hampshire municipalities, downtown stormwater drains have high concentrations of fecal contaminants, even during dry weather (NHDES, 1997; NHDES, 1998; Jones, 1998b). This suggests that sewer pipes that cross paths with the storm drains may leak contaminants into the drains. During runoff events, contaminants that accumulate in the drains are washed into the receiving waters. Thus, the system of pipes associated with municipal sewage treatment facilities may be sources of contaminants. The estimated cost for rehabilitating these systems in the coastal urban areas is well in excess of $200 million (NHDES, 1997).

2.2.2.3 Septic Systems

Many shoreline areas adjacent to the shellfish waters of New Hampshire are still served by septic systems. These systems contain high levels of bacteria and nutrients (Jones, 1998d) that can leach into groundwater. An extensive two-year study in Seabrook focused on the potential for existing, operational residential septic systems to contaminate groundwater and adjacent surface waters (Jones et al., 1996; 1995). Little evidence of significant contamination of groundwater downgradient from septic systems could be documented. At one site with a high water table, bacterial contaminants were detected ~9 meters downgradient in the groundwater. Analysis of saturated soil cores showed the presence of high concentrations (>100,000/g soil) of C. perfringens, evidence of long-term and probably cumulative contamination. Other sites also had contaminated soils at downgradient (away from the system in the direction of groundwater flow) areas. The main limitation of any study of subsurface environments is the difficulty of finding contaminant plumes without extensive exploration. The studies concluded that septic systems are indeed potential sources of contamination to tidal waters when systems are located close to the shore, especially in densely populated areas in soils with high water tables and course-grained, excessively-drained soils.

Seabrook has recently connected all residences and businesses to their new sewer system. There are still houses close to tidal waters that remain on septic systems in Hampton and Hampton Falls (NHDHHS, 1994a). The impact of disconnecting the septic systems on water quality was investigated by Jones (1997). No significant improvement in Harbor water quality was observed, possibly because the Mill Creek area had not yet been connected to the WWTF.

Septic systems are numerous around the Little Harbor area in Rye and in some areas in New Castle (Jones and Langan, 1996c). Septic systems are also common around Great and Little bays (Jones and Langan, 1995b), the Squamscott River (Jones and Langan, 1995c) and in the Oyster River watershed (Jones and Langan, 1994c; 1993a). Large areas with houses served by septic systems are also present along the coast and the Piscataqua/Cocheco/Salmon Falls River areas. Thus, septic systems are a widespread, documented potential source of contamination.

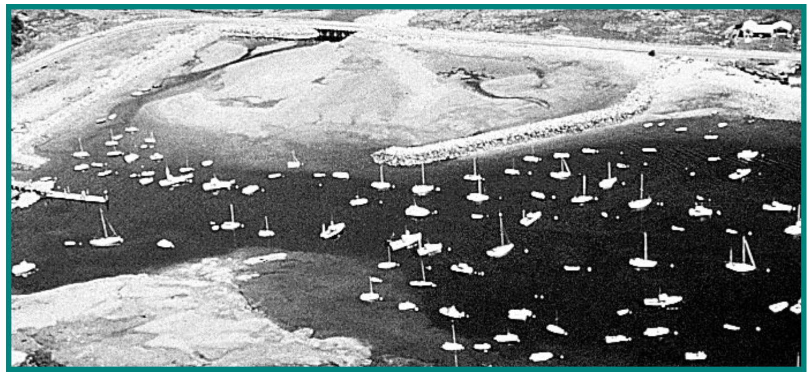
2.2.2.4 Agricultural Runoff and Other Nonpoint Sources

On a statewide basis, agriculture has not been a significant nonpoint source problem (NHDES, 1989a). The number of farms in New Hampshire and Strafford County have been declining over the past 25 years. However, horse farms are increasing. Certain activities have been problems on local levels, including manure storage and spreading practices, stable management and milk house waste management. Rockingham County Conservation District has information on contaminant runoff and management strategies for mitigating specific farm sites in the county. UNH/JEL and NHDES conducted studies at a farm in Stratham to determine the effectiveness of constructed wetlands on microbial and nutrient contaminants (Jones and Langan, 1992; 1993b). The construction of a wetland within the drainage swale between the manure storage area and the Squamscott River had no beneficial effects on contaminants during the first year after construction (Jones and Langan, 1993b). Concentrations of fecal indicator bacteria (fecal coliforms, enterococci, E. coli and C. perfringens) were all detected at elevated concentrations (> 105/100 ml) just below the manure pile, and at lower concentrations downstream. A similar trend was observed for nutrients (ammonium, nitrate/nitrite, orthophosphate).

Agricultural use of land within most growing areas have been documented (NHDHHS, 1994a; 1995; Jones and Langan, 1996c). Many of the cited farms are practicing responsible management procedures to prevent animal waste from contaminating bordering water bodies.

There are other potential sources of bacterial contamination near and within New Hampshire's shellfish waters, including storm and parking lot drains, snow dump sites, boats, wildlife and resuspended sediments. A guide for BMPs to control most potential nonpoint sources of pollution is published (NHDES, 1994c) and serves as a useful reference. NHDES has recently been successful in improving and increasing the number of coastal boat pump-out facilities. Further improvements are expected each year. Recent sanitary surveys for some coastal waters include marina assessments (NHDHHS, 1994; 1995; Jones and Langan, 1995b; 1996c).

Animal feces is often mentioned as a probable source of bacterial contamination in stormwater runoff (Jones, 1999; Oakland, 1983). In almost every case, the justification for such conclusions is that no human source could be identified, so the investigators conclude that animal waste must be the source, usually without any direct documentation. Recent studies have shown many previously unsuspected sources of stormwater contamination exist in coastal New Hampshire towns, including stormwater drains, sewer pipes, stormwater treatment systems, etc., including areas where animal feces had been previously suspected (Jones and Langan, 1996b; Jones and Langan, 1993a). More recent studies have shown underground sewage pipes contaminate stormwater drains in urban areas (Landry, 1997; Jones, 1998b). It is likely that human sources of fecal contaminants remain more significant than animal sources in New Hampshire's Seacoast (Jones, 1999). However, the issue of the source of nonpoint source pollution, whether it is of human, animal or other origin, is an extremely important question to address. Not only is it necessary for identifying the source of contamination, but it is essential for determining the public health significance of fecal contamination. A new study by NHDES and UNH/JEL will use new biotechnological methods to differentiate between human and other sources of *E. coli* isolates from New Hampshire coastal waters.



Rye Harbor

2.2.3 MODELING AND DYE STUDIES FOR BACTERIAL FATE AND TRANSPORT

Computer modeling of stormwater runoff impacts to the tidal portion of the Oyster River was conducted as part of a study by Oakland (1983). The goal was to assess impacts relative to state standards for coliform bacteria and dissolved oxygen standards, and assess effectiveness of stormwater control measure implementation. The results of the modeling confirmed observations that coliform standards would be violated routinely during storm events. Violations, even during dry weather, would be most frequent at upstream sites and during ebb tides. Dissolved oxygen standards would be violated much less frequently, only during 28% of storms. The violations would be expected to be short-lived during ebb tides only in the upper reaches of the tidal river. The model found that only Mill Pond, as a river-run impoundment, would have significant impacts on coliform loading, while vacuum cleaning of impervious surfaces could significantly reduce BOD loading.

Numerous dye studies have been conducted to determine potential contamination plumes and contaminant transport from various point sources. Ballestero (1988) reported on a field dye study and calculations for dilution and dispersion using MERGE, a contaminant plume modeling program, for the new Dover wastewater treatment plant outfall diffuser in the Piscataqua River. The purpose of the study was to determine water quality criteria for conservative contaminants in the effluent. The zone of initial dilution was set by the state to be 0.25 miles upstream and downstream from the diffuser. Average dilution at these distances was calculated to be 26,000, with significant dilution occurring as a result of the initial jet aspiration from the diffuser as the effluent entered the river. A modeling study was also conducted for a proposed diffuser for the Newmarket WWTF.

Other dye studies have been conducted to establish safety zones for shellfish harvesting around WWTFs and marinas. A recent dye study was conducted by the US EPA at the Great Bay Marina in Little Bay, but the results have not yet been published. In Hampton Harbor, a dye study was conducted to determine the safety zone downstream from the Hampton WWTF (Fugro-McClelland, 1993).

In Great Bay, the most recent sanitary survey (NHDHHS, 1995) identified the WWTFs in Durham and Newmarket as the plants with the greatest chances of impacting shellfish harvesting. There have been recent dye studies conducted at both sites, but the data are not yet published. An EPA model, CORMIX, was used to model discharges of fecal coliforms from the WWTFs (Langan and Jones, 1995a). At the Newmarket WWTF, the worst case scenario was for a release at mid-falling tide, in which case the plume would reach the mouth of the Lamprey River in 7.2 h with a concentration of 750 fecal coliforms/100 ml. The mouth of the river is an area classified as prohibited for shellfish harvesting. Thus, another model (Brown and Arrelano, 1979) was used to estimate time for the plume to reach the closest approved areas. It was estimated that the total time for the plume released at mid-falling tide to reach restricted waters is 28 h, which is sufficient for closing the area to shellfishing. At the Durham WWTF, the worst case scenario was found to be a release at high tide, in which case the plume would reach the mouth of the Oyster River in 4.2 h with a concentration of 420 fecal coliforms/100 ml. Further transport of bacteria to the Langley Island area could take a total time from a high tide release of 8-12 h.

In Hampton Harbor, CORMIX was used to model transport and survival of bacteria discharged from boats moored in Seabrook Harbor during fall-spring when the clam flats in the Harbor are open for harvesting (Langan and Jones, 1995b). Model simulations were run for both a slug release and a slow, continuous release of bacteria over a six hour time period from the vessels. The concentrations of bacteria in the plume at the edge of the adjacent clamflat for both types of releases were 13 and 0.02 fecal coliform/100 ml, respectively, which are both below the regulatory limit of 14 fecal coliforms/100 ml. The conclusion of the study was that the boats present during colder months do not pose a risk of significant contamination to adjacent clamflats. However, because boating activity increases significantly during warm months (mid-May to mid-September) it is recommended that clamflats remain closed during these times. This study did not address the Hampton Marina, which typically has many more boats than Seabrook Harbor.

Current direction and velocity measurements have been used to help predict bacterial transport and impact to shellfishing areas in Hampton Harbor (Langan and Jones, 1995b) and Little Harbor (Jones and Langan, 1996c). In Little Harbor, transport of bacteria discharged from boats at the Wentworth Marina and in the nearby mooring area to shellfishing areas were modeled using estimated discharges and current velocities and directions. Using a variety of scenarios, the modeling effort found it likely that water with fecal coliform concentrations exceeding 14/100 ml could reach clamflats under worst case conditions. Jones and Langan (1996c) recommended that shellfishing be allowed only during colder months when boat traffic and usage is negligible.

2.2.4 IMPACTS OF FECAL-BORNE BACTERIA ON SHELLFISHING

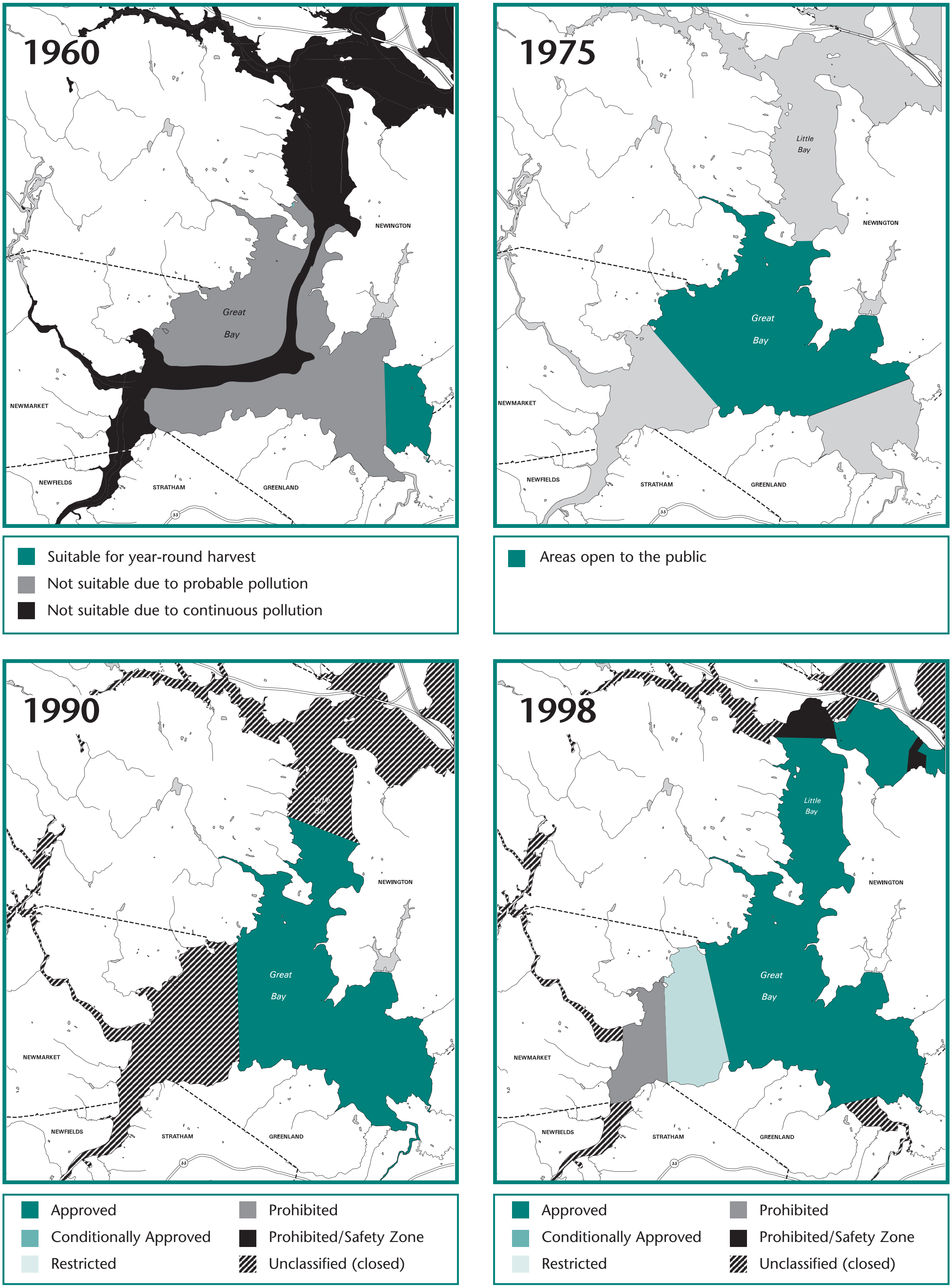
New Hampshire has abundant and valuable shellfish resources. Many citizens have enjoyed the recreational harvest of clams, oysters and mussels over the years in Great and Little bays, Hampton Harbor, Rye Harbor and Little Harbor. However, during the past few decades, all or portions of these areas have been closed for shellfishing because of unacceptable concentrations of bacterial contaminants. Much effort has been dedicated to determining which areas are safe for shellfish harvesting and how to open other areas.

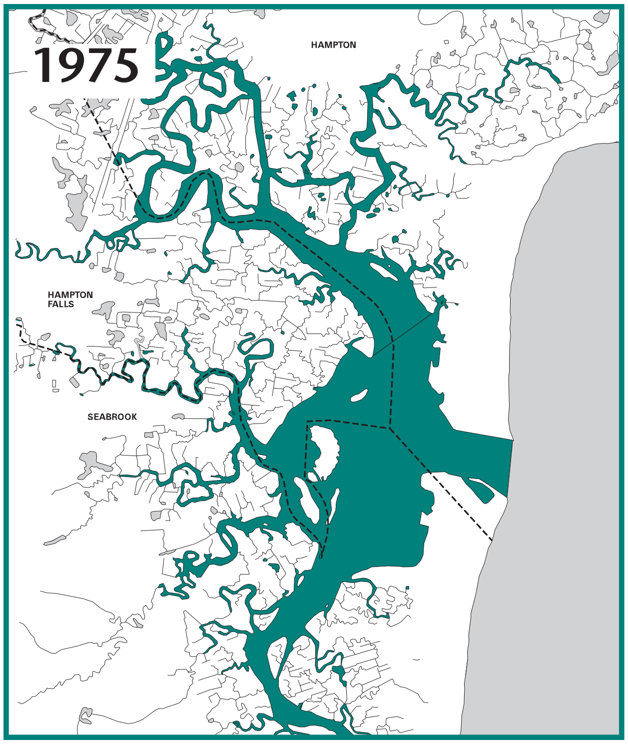
2.2.4.1 Historic Sanitary Assessments of Shellfish-growing Waters

Bacterial contamination of the shellfish growing waters of New Hampshire has been a challenging, continuous problem. New Hampshire has assessed the sanitary conditions of tidal water bodies since 1957 (NHWPC, 1960). Early data on bacterial contamination Jackson (1944) reflected the high loading of untreated sewage into the tributaries to Great Bay Estuary: every tributary had average total coliform concentrations of >800 /100 ml. Total coliform concentrations were much lower at sites in Great and Little bays, although still elevated compared to more recent data and in excess of the limit of 70 total coliforms/100 ml for shellfishing.

Early routine state assessments of the sanitary quality of tidal waters began in 1957 (NHWPC, 1960). The 1960 report included a map delineating suitability of water quality for shellfishing in the Piscataqua River/Great Bay Estuary (Figure 2.17). Only a small portion of eastern Great Bay (Greenland Bay) near the shore between Fabyan and Pierce points was classified as suitable for year-round harvest of shellfish for direct marketing. The rest of the estuary was considered unsuitable for year-round harvesting because of the continuous presence of pollution by raw sewage, except for much of the central area of Great Bay and the outer deeper areas of Portsmouth Harbor. The classification was based on only a few samples (one sample/site in some cases). By 1975, New Hampshire published shellfish waters classification maps based on a median 70 total coliform/100 ml limit for Class A tidal waters (Figures 2.17 and 2.18; NHWSPCC, 1975). Areas where median total coliform concentrations were <70/100 ml included eastern Great Bay between Nannie Island and Birch Pt. beyond the mouth of the Winnicut River, two areas near the western shoreline around the Footman and Vols Islands, the lower tidal portions of the Oyster and Bellamy rivers, Little Harbor and southern portions of the Back Channel, outer Portsmouth Harbor, the northern half of Hampton Harbor and

FIGURE 2.17 *Great Bay Estuary shellfish waters classification trends from 1960 to 1998.*





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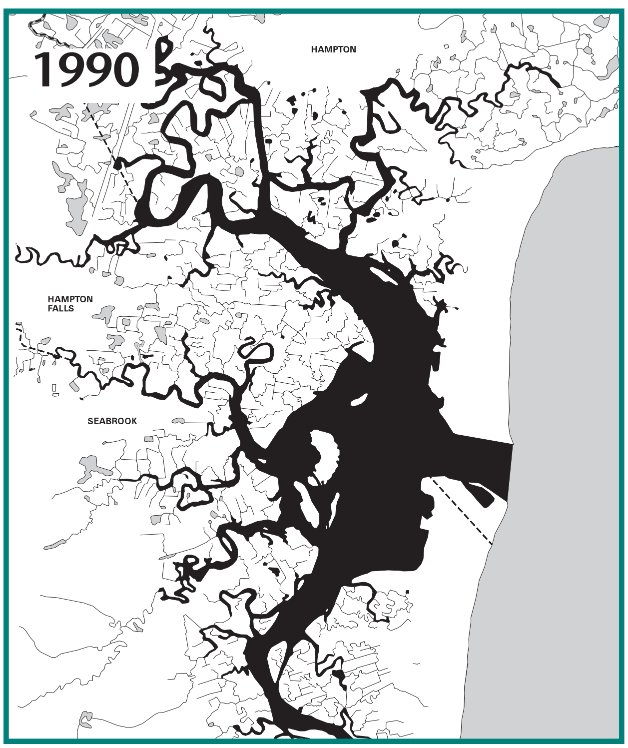
Conditionally Approved	Prohibited/Safety Zone
Restricted	Unclassified (closed)



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Conditionally Approved	Prohibited/Safety Zone
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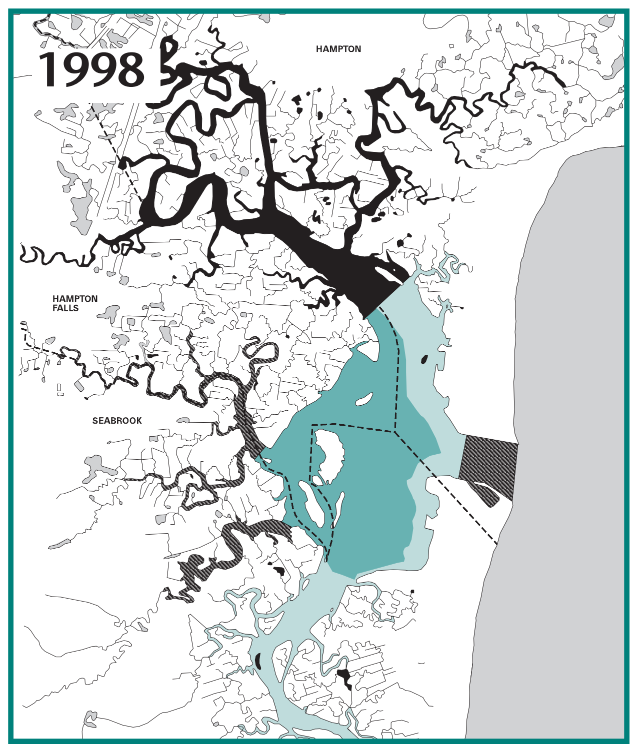


Approved	Prohibited
Conditionally Approved	Prohibited/Safety Zone
Restricted	Unclassified (closed)

lower portions of some tributaries, Rye Harbor and the whole of New Hampshire's Atlantic coast. Point sources, especially the WWTFs, were the major sources of contamination, and upgrades and construction were slated to occur within a few years of the reports for all areas not currently treating waste with the best available technology.

Contaminated shellfish waters became an even more important issue for the public and their legislative representatives after the NHDHHS closure of Hampton and Little harbors in March, 1989 (NHDES, 1989a). A Shellfish Committee was formed in March, 1988, and ensuing efforts focused on identifying sources of contaminants and eliminating them where possible. A report was written by the agency personnel on the committee in 1989 entitled "Interagency Report on the Shellfish Waters of New Hampshire" to outline what steps were needed to reopen shellfish beds. The report included a few, high priority recommendations/actions:

- prioritize the elimination of sources of bacterial contaminants and conduct a cost/benefit analysis relating remediation costs to the value of shellfish harvest activities;
- increase the effectiveness and efficiency of existing WWTF wastewater disinfection systems;
- communities should survey shorelines and eliminate nonpoint sources of pollution;
- identify sources of pollution where obvious point sources are present;
- prioritize state and federal funding to support WWTF construction and nonpoint programs in coastal communities.

The State began to make progress on each of the key recommendations soon after the 1989 Interagency Shellfish (Flanders, 1989) report was published. By 1991, improvements had been made to Dover, Exeter, Newmarket, Hampton and Portsmouth WWTFs (NHF&G, 1991). Some failed septic systems were identified and abated in Seabrook, Rye eliminated its coastal discharge of raw sewage by building a sewer line to Hampton and all but two CSOs were eliminated in Portsmouth. Shoreline surveys were conducted in Great Bay and the Bellamy River by state agencies (see below), while sources of contamination in the Bellamy River were identified and abated. Some remote residential areas in Hampton were connected into the town sewer system. For all growing areas (Great/Little Bay; Little Harbor; Hampton Harbor; Rye Harbor), specific water quality problem areas were identified, described and prioritized. Concurrent with these efforts were a number of water quality monitoring programs run by state agencies and UNH. The shellfish program continued monitoring waters to support classifications, NHDES continued monitoring some upstream areas as part of their ambient water quality monitoring program, and UNH/JEL initiated monitoring in Great Bay as part of the GBNERR program. However, the 1991 report (NHF&G, 1991) recognized the need for more extensive water quality monitoring in key areas to document improvements in water quality and to support reclassification of areas. The improvements in WWTFs and elimination of major point sources of contamination also provided conditions conducive to assessing NPS pollution.

The shellfish growing waters of Great Bay were the focus of shoreline/sanitary surveys in 1988-91: the Bellamy River (NHDES, 1991) and Great Bay (NHDHHS, 1992). The Bellamy River survey found an unpermitted pipe discharging bacterial contaminants near the Sawyer's Mill apartments in Dover near the tidal dam. No evidence of failed septic systems or other nonpoint sources of contamination was detected, and further studies were recommended. In the Great Bay sanitary survey, water samples collected along the northwest shoreline of Great Bay were all elevated (330-3,300 total coliforms/100 ml) above the total coliform limit of 70/100 ml (NHDPHS, 1992). The dominant source of contamination was considered to be WWTFs discharging into nearby tributaries.

Indigenous estuarine bacterial pathogens like vibrios have been a significant public health concern in the southern areas of the US. In New Hampshire, there has been no documented evidence of food poisoning or wound infections in the local communities associated with the incidence of any *Vibrio* sp., except for an incident of *V. parahaemolyticus* gastroenteritis resulting from consumption of oysters taken from Great Bay waters that occurred in June, 1992 (Dr. R. Rubin, personal communication).

2.2.4.2 Present Conditions

A recent sanitary survey in Great Bay was conducted (NHDPHS, 1995; Jones and Langan, 1995b). The approved area was expanded northward in Little Bay from the cable crossing (Figure 1-6) based on monitoring at NHDHHS stations (Figure 2-3). The northern boundary for the approved area now extends from Fox Point (43°07'10" N. Latitude, 70°51'35" W. Longitude) to the western shore of Little Bay at Durham Point (43°07'14" N. Latitude, 70°52'10" W. Longitude). A new sanitary survey and related studies have focused more intensive monitoring in lower Little Bay and the Bellamy River (NHDHHS, 1998; Jones, 1998a). The shoreline survey and fecal coliform concentrations at five of the six sites were consistent with an approved classification of much of lower Little Bay. Initially, only an area around Broad Cove was classified as approved, as other areas required additional samples. In 1998, most of the rest of lower Little Bay was re-classified as approved, except for an area from the mouth of the Oyster River east to Fox Point, and areas around the two marinas. In Great Bay, a restricted area has been established in the southwestern corner of Great Bay toward the mouths of the Lamprey and Squamscott rivers. The classification of eastern Great Bay has been clarified and is almost all approved, except Greenland Bay south of a line extending from Pierce Point west to the Greenland shoreline.

Little Harbor was the focus of a preliminary sanitary survey in 1995-96 (Jones and Langan, 1996c). Water quality was found to meet approved classification standards in Little Harbor, and no significant sources of pollution were documented. The Wentworth Marina was considered to be a significant potential source of bacterial contaminants. A pumpout facility replaced in 1997 using Clean Vessel Act support and private funds. Even though it has pump-out facilities that are extensively used, such large marinas are regarded as potentially significant sources of contamination relative to classifying shellfish areas. The statewide closure of shellfishing during warm months, June through early September (November for Hampton Harbor), coincides with the timing of the greatest use of the marina, mid-May through mid-September. The absence of boaters at the marina during colder months resulted in little impact of the marina on water quality (Jones and Langan, 1996c), and would probably not be a concern if the area was opened during cold months for shellfishing.

In the rest of the Little Harbor area, the Witch and Seavey Creek area has some problems with water quality and further studies are needed to identify sources. The Back Channel area should also remain closed because of the CSOs in Portsmouth and other recently identified sources.

A sanitary survey was conducted in Hampton Harbor during 1993-94 to support reclassification of the closed shellfish waters (NHDHHS, 1994). The study involved intensive water quality monitoring, experiments designed to test a variety of conditions and consideration of all potential and known pollution sources. The effort resulted in reclassification of portions of Hampton Harbor to "conditionally approved", limited by rainfall events and closed during warm months (June-October) because of the increased summer population. The classification was based on sampling at NHDHHS sites (Figure 2.12). Elevated concentrations of fecal coliforms at a few sites in the harbor near the mouth of Mill Creek and near River St. and Cross Beach Rd. were investigated further in 1995 (Langan and Jones, 1995a & b). The study and a newer study (Jones, 1997) suggested that elevated bacterial concentrations may originate from Mill Creek or possibly from resuspended sediments; no clearly defined sources were found. Improved water quality in recent years has resulted in a recent upgrading of the shellfish harvest classification of the large Middle Ground clam flat in Seabrook from restricted to conditionally approved (NHDHHS,1998). Clamming can occur from November to May except after rain events of ≥0.1 inches of rain in 24 hours. In addition, the rainfall condition of approved classification has been modified to be seasonal, with less restrictive conditions (0.25" rain per 24 h) in effect for all areas during December through March. It is hoped that complete disconnection of all septic systems in the area will result in improved water quality so even more clam flats can be opened.

2.2.5 MICROBIAL CONTAMINATION Impacts on Swimming and Other Recreational Uses

There have been no reported incidences of water-borne disease in New Hampshire at least since 1992 (NHDES, 1994a; 1996b). Microbial contaminants would be a concern at bathing beaches if swimmers ingested water and became ill. Bacterial indicator standards are based on USEPA studies of disease incidence in association with swimming. Thus, the enterococci standard for tidal recreational waters was developed to protect humans from fecal-borne pathogens. The data from the NHDES 305(b) reports showed swimming was only restricted at open ocean sites in 1991-1994 and at a coastal shoreline site from 1988 to 1990.

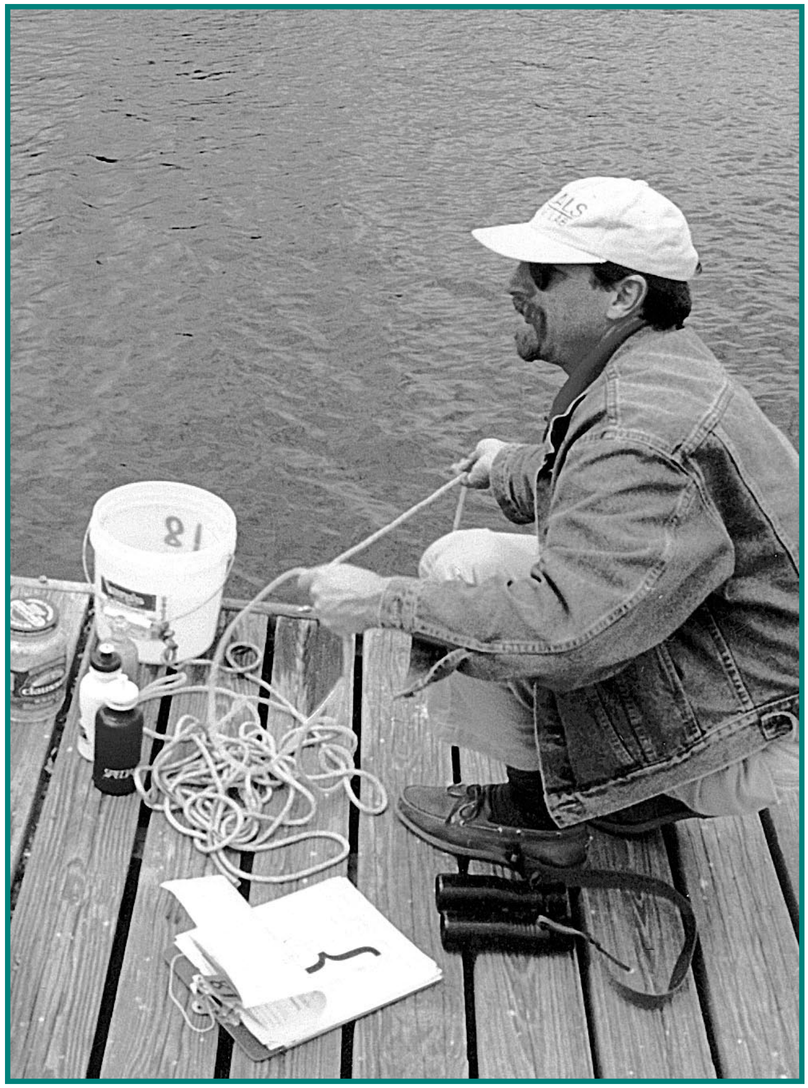
Some temporary closures of beaches in New Hampshire occur during warm months when beaches become overcrowded. The heavy population of swimmers can cause concentrations of fecal-borne bacteria to be present at levels that exceed standards, and time is needed for the water to become clean again prior to re-opening beaches.

2.2.6 FECAL-BORNE PATHOGENS Historical Studies on Indicators and Pathogens

Historically, there has been a great deal of research in Great Bay conducted by researchers at the Jackson Estuarine Laboratory and the Department of Microbiology at the University of New Hampshire on the various aspects of microbial pathogens. The estuary has served as a useful site to conduct these studies, as sewage discharges have contaminated shellfish-growing areas for a long time (NHWPC, 1960; NHWSPCC, 1975; 1981). Slanetz et al. (1964) found good correlations between membrane filtration and multiple tube fermentation tests for coliforms in shellfish and water, and showed that not all positive fecal coliform tubes contained *Escherichia coli.* Fecal streptococci and fecal coliforms were useful indicators of fecal pathogen contamination, as *Salmonella* sp., and on two occasions, Coxsackie viruses were detected in shellfish and waters from areas having high levels of fecal indicator bacteria (Slanetz et al., 1968). However, *Salmonella* sp. (Slanetz et al., 1968) and enteric viruses (Metcalf et al., 1973; Metcalf, 1975) were also detected in samples of water and oysters from areas that met the coliform standard for approved shellfish-growing waters. One general conclusion of the historical studies was that enteric viruses and *Salmonella* sp. had a greater ability to survive than indicator bacteria in estuarine environments, and that these pathogens were often associated with irregular introductions, or pulses, of contamination into the estuary. The findings provided early evidence that contributed to growing doubts about the adequacy of using total coliforms for classifying approved shellfish waters, especially with low indicator levels. The occurrence of the specific pathogens *Salmonella* sp. and enteric viruses was never correlated with any reported incidence of disease caused by these microorganisms in surrounding communities.

The sources and fate of microbial contaminants in Great Bay were the subject of further studies. Metcalf and Stiles (1968) found that enteric viruses were discharged from sewage effluent pipes and disseminated throughout the estuary. The viruses were rapidly taken up by oysters and retained for months within shellfish, especially during cold winter months. Introduction of chlorination as treatment of sewage by a municipal facility caused dramatic decreases in coliform, *Salmonella*, and enteric virus levels, although the pathogens could still be detected in treated effluent on occasion. Slanetz et al. (1972) found rapid die-off of indicator bacteria in oxidation ponds at three wastewater treatment facilities in the estuarine system, especially when three to four ponds in succession were used to treat wastewater. However, *Salmonella* and enteric viruses could be isolated from all ponds, especially in cold (1-10°C) water. Such findings are important relative to the oyster harvest season in Great Bay, which spans the cold autumn through spring months and is only closed during the warm summer months. More recent studies on pathogens in oysters from the Piscataqua River showed no detectable *Salmonella* sp. in shellfish prior to processing at a commercial shellfish depuration facility in Maine (Jones et al., 1991).

Presently accepted methods for detecting enteric viruses are too expensive, slow, and complex to be adopted for routine analysis of water and shellfish. However, more rapid and precise methods for detecting enteric viruses are being developed at UNH. For example, application of radioactively labeled cDNA probes for poliovirus and Hepatitis A virus showed the presence of these viruses in shellfish and water from closed areas in Great Bay (Moore and Margolin, 1993; Margolin and Jones, 1990; Margolin et al., 1990). Gene probe assays showed good agreement with traditional tissue culture methods for virus detection. Comparison of virus incidence with levels of bacterial indicators in the Oyster River revealed no clear trends. Levels of bacterial indicators were consistent with the classification of the river as prohibited for shellfishing, but showed little relationship to the presence or absence of enteric viruses.



A. REID

Water quality sampling

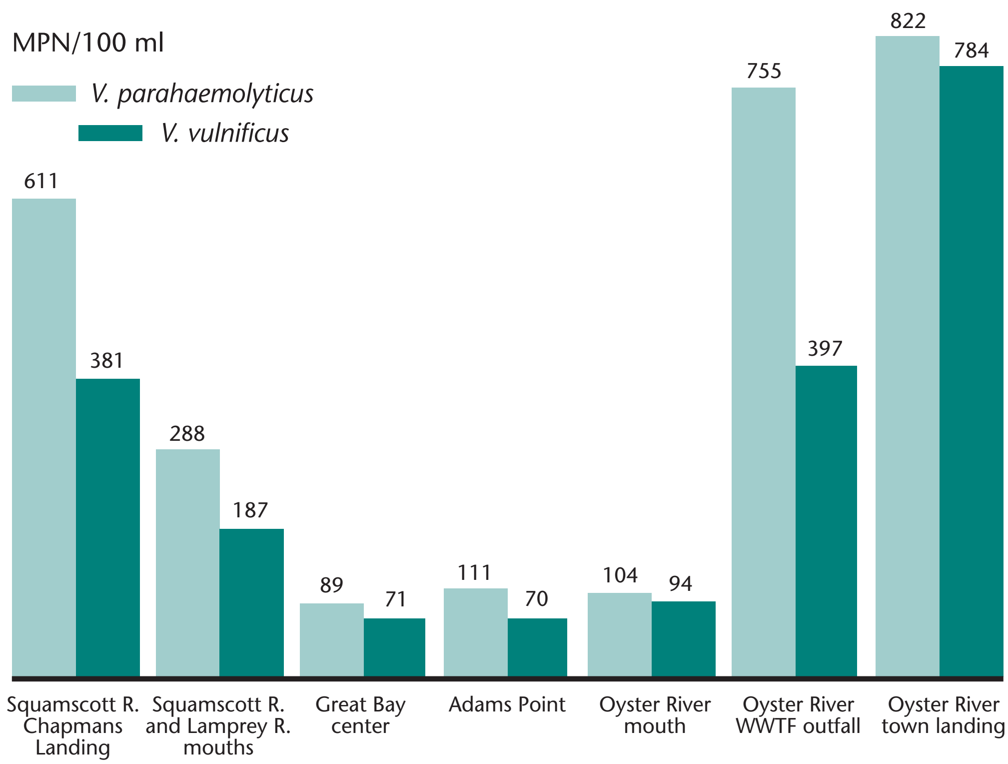
An ongoing study is focusing on viral contamination of groundwater in northern New England (D. Heath, personal communication). Total culturable enteric viruses and PCR analysis of poliovirus, hepatitis A and Norwalk virus are being measured in comparison to other microbial indicators and dissolved nutrients. Groundwater samples are being collected from drinking water wells located in close proximity to septic systems and that have had past contamination problems.

2.2.7 AUTOCHTHONOUS MICROBIAL PATHOGENS

Non-fecal bacterial pathogens that are indigenous to and common inhabitants of estuarine environments are also potential health hazards. In particular, the *Vibrionaceae* have been associated with shellfish-borne disease incidence and wound infections resulting from exposure to marine waters (Rippey, 1994). Bartley and Slanetz (1971) found *Vibrio parahaemolyticus* in oysters and estuarine water from Great and Little bays in September and at decreasing levels through November. *V. parahaemolyticus* has also been detected in oysters (Jones et al., 1991) and water (Jones and Summer-Brason, 1998; Summer-Brason, 1998; Jones et al., 1997) from the Estuary in more recent studies. Another vibrio, *V. vulnificus*, was detected in 1989 for the first time north of Boston Harbor in the Maine and New Hampshire waters of the Great Bay Estuary (O'Neill et al., 1990). This discovery did not necessarily mean that it was a new inhabitant of the estuary. Many other reasons are related to why it had not been previously detected, including no one had tried to detect it, it was only recognized as a bacterial species in the late 1970s and there was no incidence of *V. vulnificus*-related disease to cause alarm. It has since been detected routinely in all of the tidal portions of the major tributary rivers of the estuary, where shellfishing is not permitted, but detection is extremely rare and at low concentrations in the areas of Great Bay open to shellfishing (Figure 2.19; Jones et al., 1997; O'Neill et al., 1990; Jones et al., 1991). A relatively high incidence of hemolysin-negative, or potentially non-virulent strains of *V. vulnificus* have been isolated from the estuary (O'Neill et al., 1991).

More recent studies in Great Bay and the Oyster River helped to delineate the ecology of *V. vulnificus*. This is important for prediction of conditions that may result in higher concentrations of the organism and for developing post-harvest processing strategies for eliminating

FIGURE 2.19 *Geometric mean* Vibrio vulnificus *and* Vibrio parahaemolyticus *concentrations at low tide (MPN/100 ml) in Great Bay Estuary by site during June-September, 1993-95.*



TOXIC ORGANIC AND METAL CONTAMINANTS

Numerous historical and current studies have focused on organic contaminants, metals and metalloids in coastal New Hampshire, especially in Great Bay. The major sources of information can be found in reports from the 1991-93 ecological risk assessments for the Portsmouth Naval Shipyard, the Gulfwatch 1991-98 annual reports, the Army Corps of Engineers dredge project data, NPDES monitoring data, numerous reports by Normandeau Associates, reports from the former Pease AFB, and scientific papers from a few UNH laboratories in the departments of Chemistry, Earth Sciences and Microbiology. Numerous other studies conducted by private firms, the University, and both state and federal agencies also provide important information. Contaminants that have the most available information include chromium, mercury, tin and lead, based on their local distribution, historical and current sources, potential toxicity and scientific interest.

Small scale, light manufacturing is practiced in Portsmouth along the Piscataqua River and in many of the municipalities bordering the Great Bay and Hampton/Seabrook estuaries. There are no industrial activities on the shores of some coastal areas, such as Little Harbor. Other areas like the Portsmouth Naval Shipyard and Pease AFB have been the sites of significant historical storage and use of toxic contaminants. An environmental assessment of the shipyard and surrounding estuarine habitats has shown elevated levels of some toxic compounds in depositional areas and some biota (NCCOSC, 1997). Little evidence of actual toxic effects on biota was apparent. The urban areas in the coastal region have had a variety of industrial activities that have contributed unknown quantities of contaminants to surface waters over the last three centuries.

Studies have been conducted to determine the concentrations of contaminants in sediments, in organisms and in the water column, with some focusing on their effects on organisms. Information on the status and trends of toxic contaminants in these environmental compartments is presented below.

2.3.1 STATUS AND TRENDS FOR CONTAMINANTS IN WATER

Lyons et al. (1976) studied trace metal discharges into the Great Bay Estuary in the mid-1970s. Measurements were made of dissolved and "environmentally available" Fe, Mn, Cu, and Cr. Only Cr was present at levels in excess of the range found for other northern New England river systems. The data indicated a reduction of inputs to the estuary from industry compared with what had occurred in the previous decade. Scattered small projects involving analysis of tidal waters have also occurred. For example, water from the Taylor River in the Hampton/Seabrook Estuary was analyzed for nine metals and ten organic contaminants during 1985 (ESI, unpublished data). Nelson (1986) reported the analysis of water from four areas in the Great Bay Estuary for lead concentrations, which ranged from <0.05 to 0.14 mg/l.

More recent studies on contaminant concentrations in water have been conducted as part of the Portsmouth Naval Shipyard studies (Johnston et al., 1993). Initial measurements of metals in the Piscataqua River encountered problems, but samples of seep water from sites near suspected sources showed elevated concentrations of Pb, Hg, Zn, Cr and Cu, some of which may have been associated with suspended sediments inadvertently included in the samples.

Further sampling of the river and seep waters were conducted as part of the second phase of the project (NCCOSC, 1997). The data, when compared to Water Quality Criteria (WQC) for protection of both human health and aquatic life, showed measured contaminant concentrations except for copper were >10x lower than the marine chronic WQCs. All sites had copper concentrations ~10x lower than the 3.1 mg/l WQC with the highest concentration in the upper Great Bay Estuary of 0.49 mg/l, which is only ~6x lower.

NHDES measured concentrations of Al, Cu, Zn and Pb that exceeded standards in water samples from urban areas in the Lamprey River (NHDES, 1994b). They compared concentrations from samples in 1987-92 at rural sites with samples from 1992 and 1993 at urban sites. The results indicated that the metals were present at concentrations higher than elsewhere in New Hampshire. The report recommended more intensive monitoring for metals in the Lamprey River and in other rivers to help put the results into a broader context. In addition, toxicity assessments in trouble areas were also recommended. In follow-up studies, the NHOSP found Al, Zn and Cu concentrations in water samples from the Exeter River to be greater than state standards at many sites during storm events (NHOSP, 1995a), and frequent exceedences for Pb, Zn and Cu during storm events at numerous sites in the Oyster River watershed (NHCP, 1996). Elevated concentrations of trace metals in stormwater runoff in Dover and Exeter have been measured, especially during significant storm/runoff events (Jones et al., 1999).

It appears that tributaries to estuarine waters have storm-related problems with trace metal contamination. In addition to their impact in the freshwater tributaries, the contaminants potentially may be transported to estuarine waters and pose risks to estuarine biota. The high copper concentrations in the tributaries and in the upper Great Bay Estuary are good evidence that transport is occurring.

2.3.2 STATUS AND TRENDS FOR CONTAMINATED SEDIMENTS

Many studies have focused on contaminants in sediments in coastal New Hampshire. Recent efforts are providing an update to many areas not surveyed since the 1970s (Bonis and Gaudette, 1998). A comprehensive database for contaminated sediments in coastal New Hampshire areas has been compiled by the USGS and will soon be available on CD and through the Internet (Buchholtz ten Brink et al., 1994 & 1997). Data from the PNS estuarine ecological risk assessment (Johnston et al., 1994), the Army Corps of Engineers dredging projects (NAI, 1994) and various scientific papers, consulting firm reports and theses are included. In all, the database includes data for 199 samples from New Hampshire, 452 samples from Maine and 993 samples from USACE permit applications and federal navigation projects. Information in the database is from reports and papers dating from 1973 to 1994, providing the opportunity in the future to determine trends for sediment contaminants at specific sites. The data, along with data from the rest of the Gulf of Maine, are presently being validated and interpretive maps are being produced.

The trace metal at highest concentration in New Hampshire's estuarine sediments is chromium. The range of chromium concentrations in sediments is 12-2300 mg/l. The highest chromium concentrations are found in the Cocheco River, where tannery waste with high levels of chromium were discharged. Chromium concentrations in Cocheco River sediments are commonly greater than the ER-M of 145 mg Cr/l. Chromium from the Cocheco River has been transported throughout the estuary (Capuzzo and Anderson, 1973).

Examples of the latest draft versions of the USGS maps for New Hampshire are presented in Figures 2.21-23 for mercury, lead and chromium, along with an example map of lead concentrations in the US portion of the Gulf of Maine (Figure 2.24) to provide a regional perspective to New Hampshire data. Data and maps are also available for nickel, cadmium, zinc, copper, phenanthrene, fluoranthene and pyrene in both the Gulf of Maine and in the Great Bay Estuary. The three example maps presented are useful to see general patterns in contaminant concentrations. The data are comprehensive and do not distinguish between older and newer data, analytical methods, sampling methods, or sample replication. Validation of data and maps is ongoing, along with the

FIGURE 2.21

Mercury concentrations in sediments in coastal New Hampshire waters: 1973-1994.

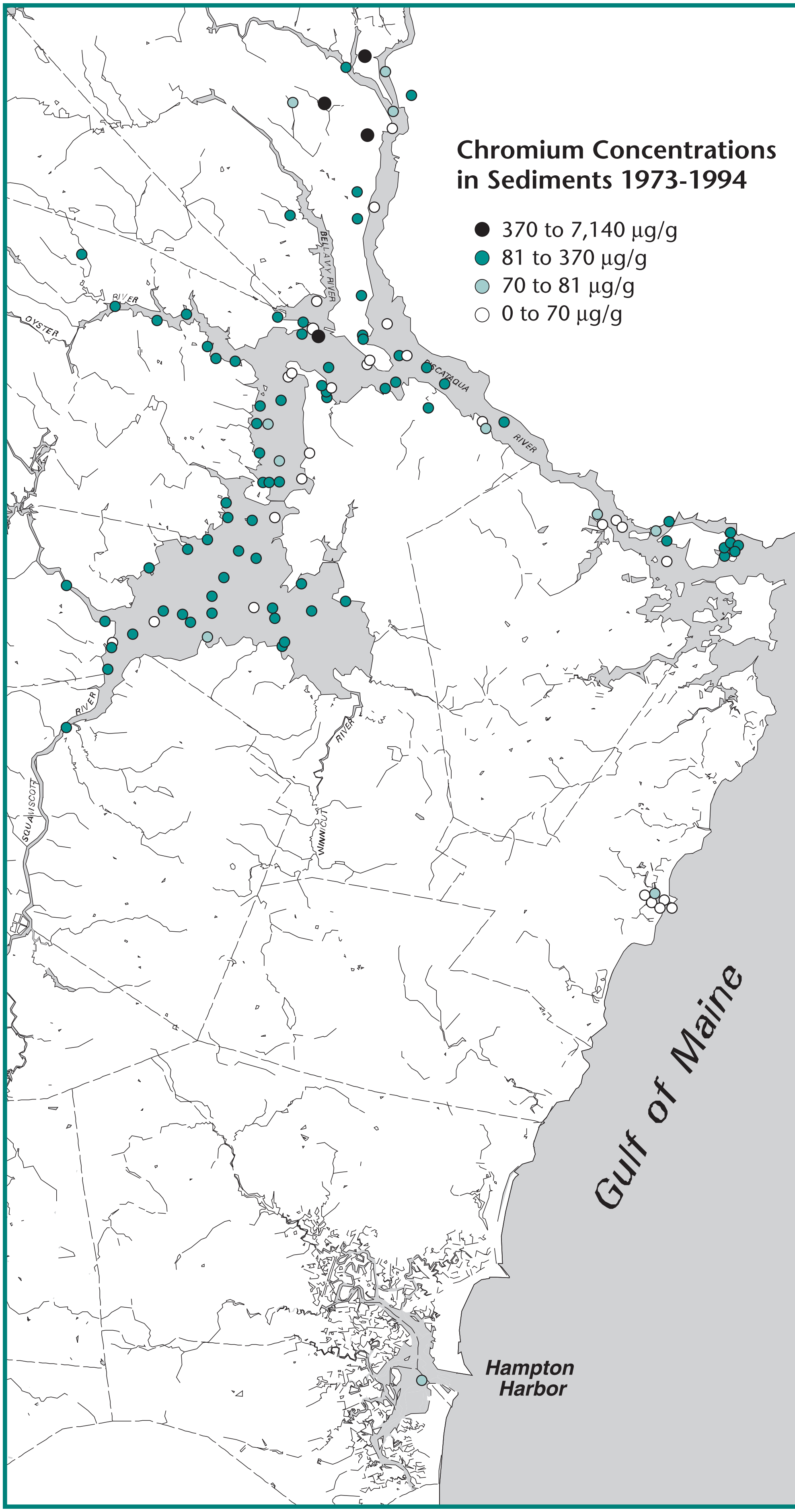
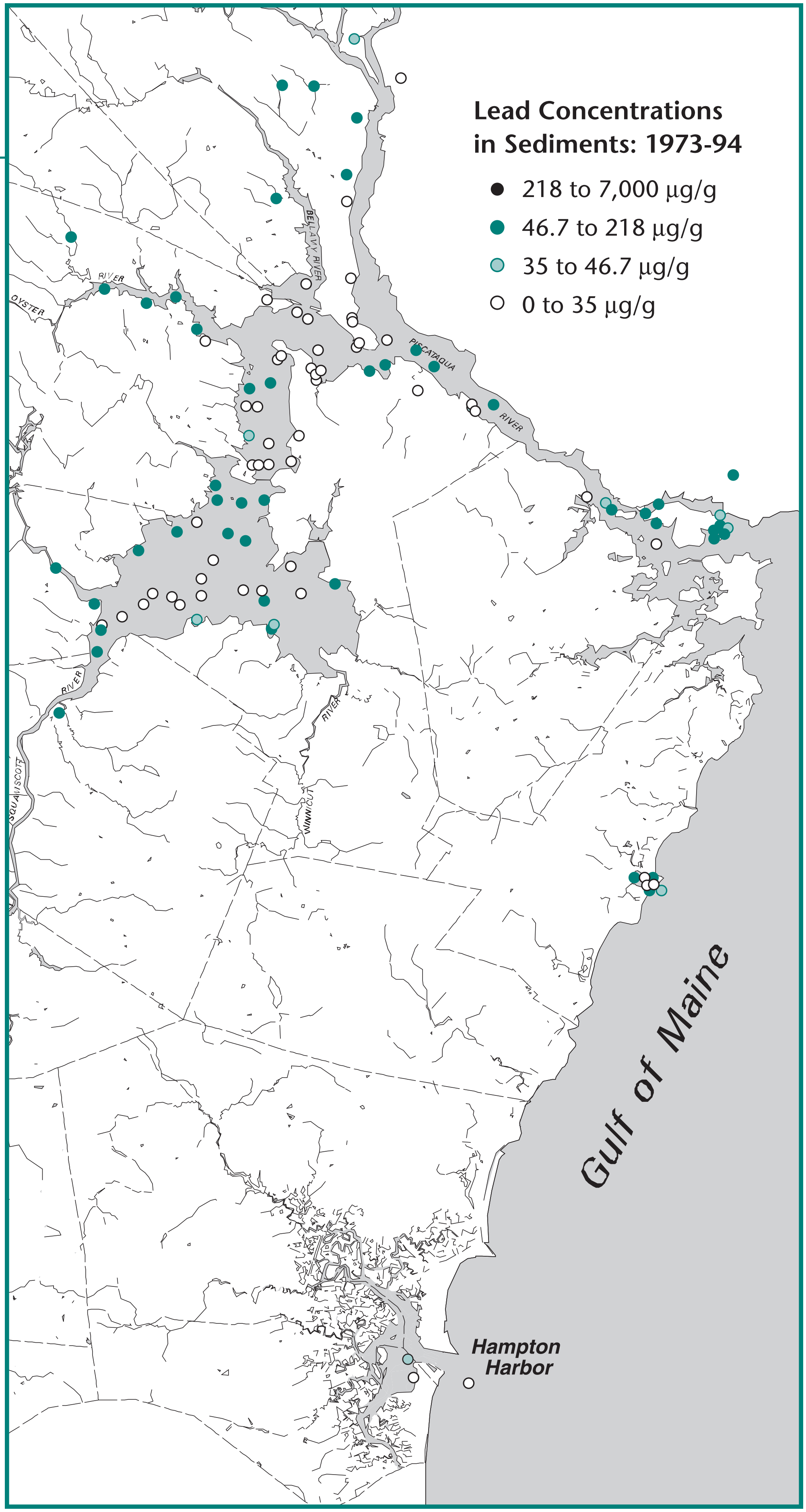


FIGURE 2.22

Chromium concentrations in sediments in coastal New Hampshire waters: 1973-1994.

FIGURE 2.23

Lead concentrations in sediments in coastal New Hampshire waters: 1973-1994.



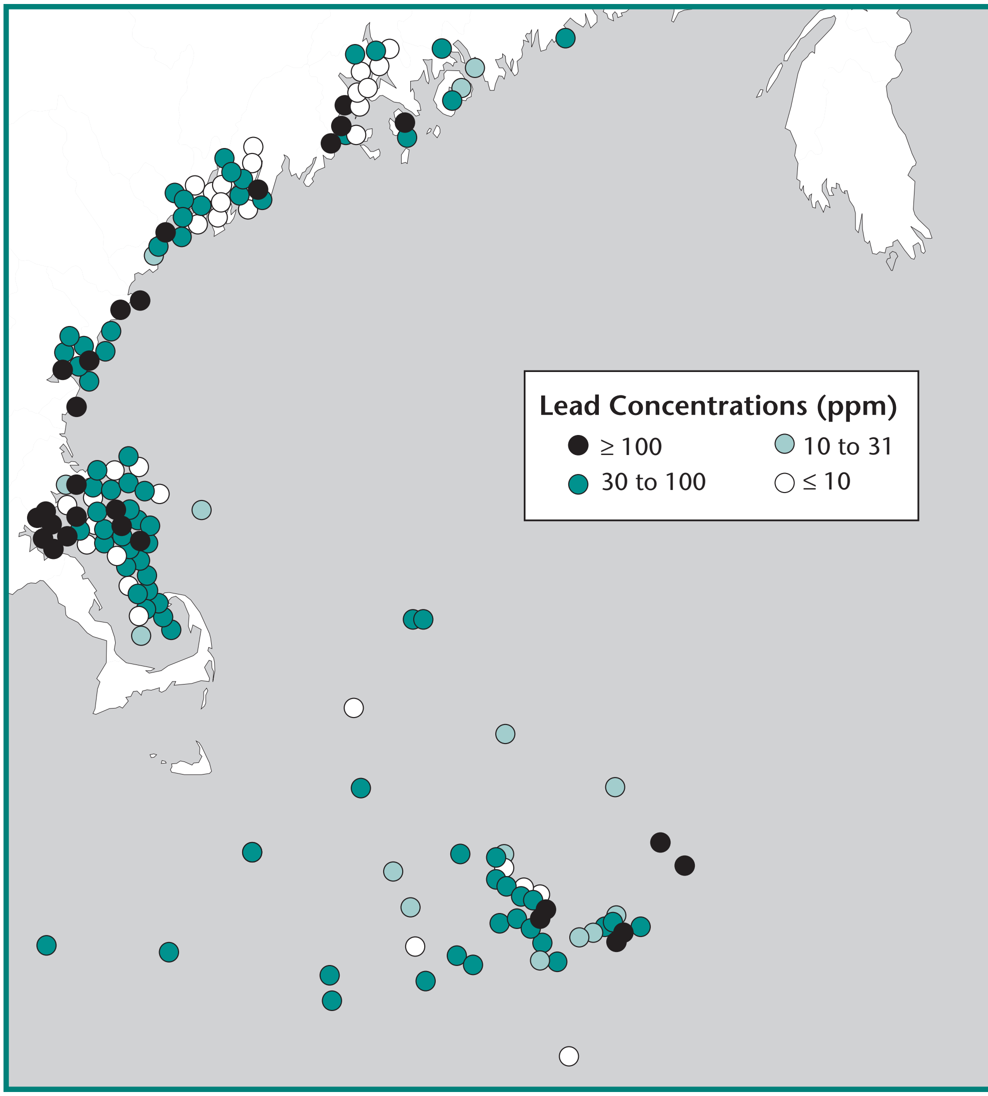


FIGURE 2.24

Lead concentrations in sediments in the U.S. portion of the Gulf of Maine and Georges Bank.

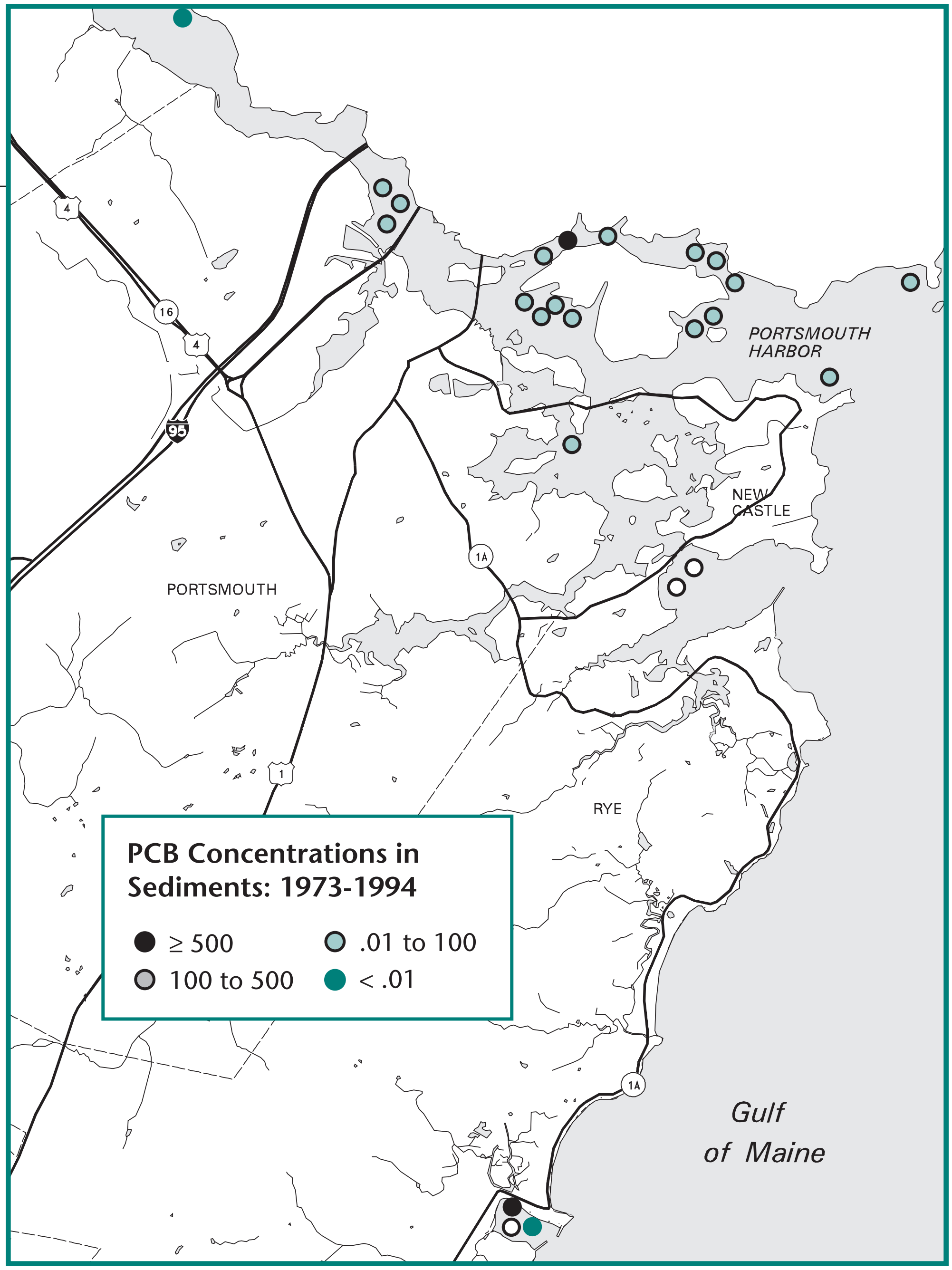
databases for organic contaminants and sediment texture.

Figure 2.21 shows numerous sites in the lower Piscataqua River and Rye Harbor that have Hg concentrations that exceed the ER-L sediment quality criterion of 0.15 µg/g (Long and Morgan, 1990), but no sites that exceed the ER-M criterion of 1.3 µg/g. The upper Great Bay Estuary generally had lower levels of mercury. Sites with lead concentrations that exceed the ER-L criterion of 35 g/g are numerous and spread throughout the entire coastal New Hampshire area (Figure 2.22). Three sites had lead concentrations greater than the ER-M level of 110 µg/g. The sites were near Seavey Island in Portsmouth Harbor and in the Squamscott River. Many sites with lower concentrations (<31 µg/g) were concentrated around Adams Point and Little Bay areas. Only four sites had concentrations of copper at or near the ER-L concentration of 70 µg/g. The sites included the same two sites that had high lead concentrations near Seavey Island, and two other sites in Great and Little bays. Relatively high (>81 µg/g) chromium concentrations are spread throughout the Great Bay Estuary (Figure 2.23), with the highest concentration in the Cocheco River. The Gulf of Maine map presents lead concentration in relation to background concentrations (20 µg/g), with values up to 2-3 orders of magnitude greater than background (Figure 2.24). Only one site (near Seavey Island) had a concentrations as high as 2.5 orders of magnitude greater than background.

As a means of assessing the impact of oil spills on sediments, sediments were collected monthly at 24 intertidal and subtidal sites throughout the Great Bay Estuary and analyzed for hydrocarbons

FIGURE 2.25

PCB concentrations in sediments in coastal New Hampshire waters: 1973-1994.



(Nelson, 1982). Nelson (1982) reported the results of analyses for PAHs and alkanes for February, 1981 at both intertidal and subtidal sites at eight different stations. Concentrations were reported for 13 different PAHs, ranging from 0 for numerous PAHs to >1000 mg/g sediment for chrysene and benzo[a]anthracene at Nobles I., Cedar Pt., Royall's Cove and Fox Pt. Alkane analysis was reported as concentrations for even and odd-numbered carbons in chains ranging from 14 to 32 carbons. Total alkane concentrations ranged from 707 ng/g sediment to 24,960 ng/g sediment. Sites with the highest concentrations included Rollins Farm (>14,800 ng/g), Broad Cove (>17,000 ng/g) Royall's Cove (>24,900 ng/g) in either intertidal or subtidal sites. Evidence of contamination from oil spills was evident at all sites, suggesting that oil spilled mainly in the lower estuary is likely transported to the upper estuary.

Dredge materials in New Hampshire have been disposed of in intertidal, nearshore, open water, upland or unknown locations (NAI, 1994). Much of the material dredged was disposed of at the Cape Arundel open water site. Some of the Rockingham County material was

2.3.3 SOURCES OF TOXIC CONTAMINANTS

Current industrial discharges of toxic contaminants are significantly less than the historical discharges that are probably the cause of much of the existing contaminants in New Hampshire sediments. Most current sources of toxic contaminants are suspected to be more diffuse sources such as urban stormwater runoff, atmospheric deposition, oil spills, and runoff plus groundwater infiltration from Superfund sites, golf courses and landfills. Stormwater runoff is the most frequently cited existing source of toxic contaminants in coastal New Hampshire (Jones et al., 1999). Stormwater runoff and associated storm event effects may also enhance contamination for some of the other sources of contaminants detailed below.

2.3.3.1 Stormwater Runoff

Stormwater runoff is the most frequently cited existing source of toxic contaminants in coastal New Hampshire. Significantly elevated concentrations of aluminum, lead, copper and zinc have been documented in freshwater tributaries (NHDES, 1994; see Status and Trends of Contaminants in Water section). Much of the stormwater and associated contaminants probably enter surface waters via stormdrains in urban areas (Jones et al., 1999; Jones, 1998b; Landry, 1997). This is currently the focus of a study supported by the NHCP. Stormwater is also suspected to enter the Great Bay Estuary directly through various streams and brooks throughout each bordering town. The area around the former Pease Air Force Base (PAFB) has been well documented. There are two drainage streams in Newington that are permitted NPDES outfalls, both formerly used by PAFB and presently used by the Pease International Tradeport (Figure 2.26). Flagstone Brook flows north from the site and eventually discharges into lower Little Bay (Tricky Cove) while McIntyre Brook flows from the runway into southeastern Great Bay. Both brooks are used for disposal of "stormwater runoff from airport activities" according to the NPDES, EPA-issued permit. Activities resulting in the production of this waste include aircraft maintenance, aircraft fueling, painting and stripping, aircraft washing and most significantly, aircraft de-icing. McIntyre Brook has the potential for having a more direct impact on the growing area than Flagstone due to the location of the discharge relative to shellfish resource areas. Major effluent characteristics that require monthly monitoring in McIntyre Brook include pH, oil and grease, primary deicing chemical, surfactants, trichloroethylene (quarterly), and total recoverable iron and zinc. Most of the runway and aircraft parking apron, industrial shop area and the entire flightline area drain into McIntyre Brook. There is an oil/water separator located near the origin of McIntyre Brook and a newly installed separator on Flagstone Brook. One of the main concerns with McIntyre Brook has been the propylene glycol content in the discharged water. This product is used in deicing aircraft and can potentially decrease the amount of dissolved oxygen in water. In 1992, as a part of the Air Force Installation Restoration program, shellfish tissue analysis was performed on samples collected in the vicinity of the Air Force Base. In an effort to evaluate the potential impacts of contaminants released from the Air Force Base into McIntyre Brook, American oysters, soft-shell clams, ribbed mussels and mummichogs were collected at the mouth of the brook where it discharges into Great Bay. Results of these analyses concluded that aluminum, arsenic and potassium concentrations in shellfish tissue samples exceeded background concentrations. However, the presence of these metals and the concentrations in which they were detected, do not pose a significant health risk to humans and were not concluded by the NHDES to be potential health risks.

In addition to McIntyre and Flagstone brooks, there are two non-permitted drainage brooks located on the Pease International Tradeport property which drain into the southeast portion of Great Bay. They are Peverly Brook and Picker-

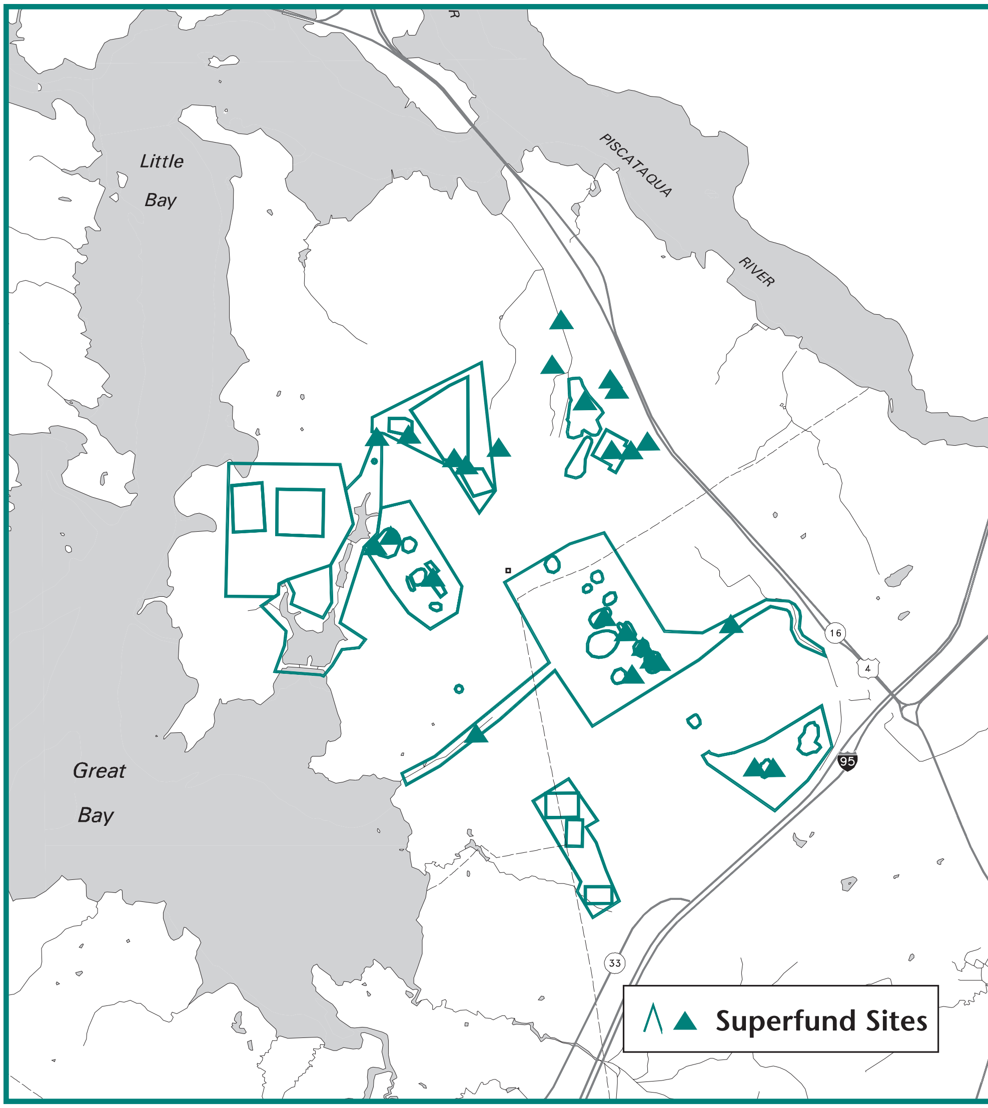


FIGURE 2.26

Superfund sites and surface waters in the former Pease Air Force base.

ing Brook. Runoff is characterized predominantly by overland flow to these streams. The Pease International Tradeport has adopted a Stormwater Best Management Practices Plan in order to properly handle all stormwater waste originating at the facility.

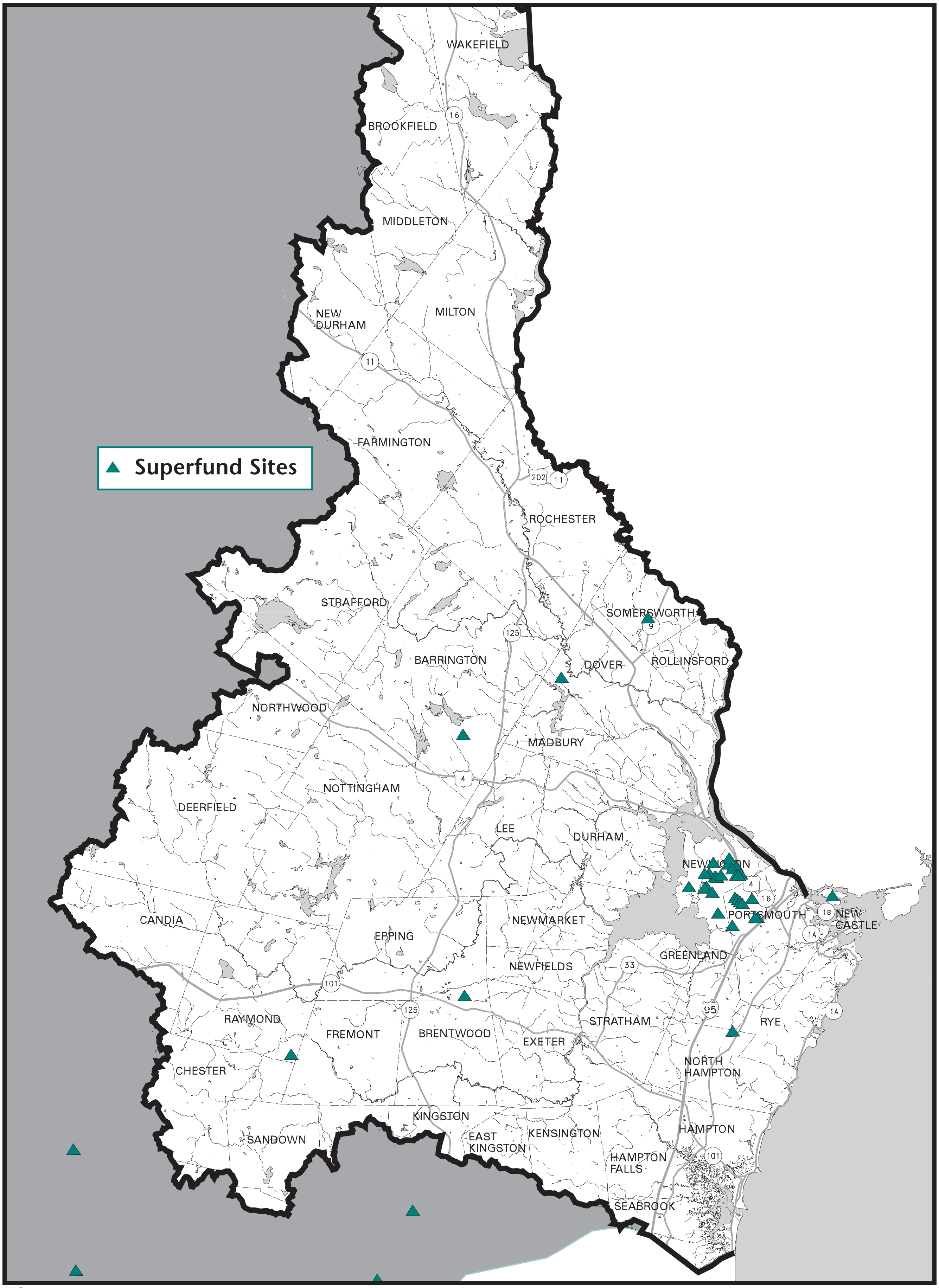
A joint UNH-JEL/NHDES study on stormwater control systems in the coastal area assessed the effectiveness of the systems to remove Al, Cd, Cu and Zn (Jones and Langan, 1996b). Concentrations of Al, Cu and Zn in the effluent from all of the systems exceeded the New Hampshire acute water quality standards for protection of aquatic life (NHDES, 1996b) during at least one storm event, especially during storms that occurred in winter. Cadmium concentrations rarely exceeded the acute standard, and exceeded the chronic standard less frequently than for other metals.

2.3.3.2 Superfund Sites

There are Superfund sites in coastal New Hampshire (Figure 2.27) with the Portsmouth Naval Shipyard, the former Pease Air Force base and Coakley landfill being of most concern to estuarine environmental quality. Copious amounts of information have been generated on environmental concentrations of contaminants, cleanup strategies, and toxicity to biota for both the Portsmouth Naval Shipyard (NCCOSC, 1997; Johnston et al., 1994) and the former Pease Air Force Base (Earth Tech, 1995). A large number of studies for these sites have been reviewed and synthesized (NCCOSC, 1997; Earth Tech, 1995).

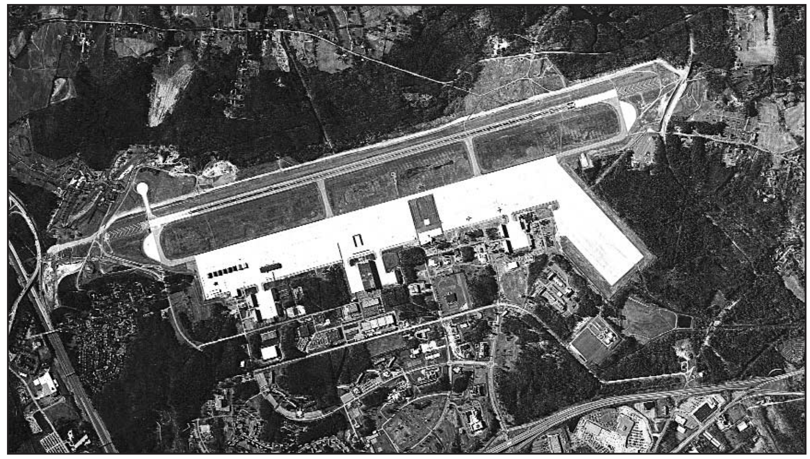
At PAFB, elevated concentrations of contaminants have been found in the sediments of some small streams, in groundwater plumes, in some biota, and

FIGURE 2.27 *Superfund sites in the coastal region of New Hampshire.*



in soil (Weston, 1992), mostly in close proximity to known sites of hazardous waste storage, disposal or discharge. Extensive measurements of contaminants in surface water, sediments and fish have been made (Weston, 1992). In addition, extensive analysis of surface water at two small rivers and sediments at three wetlands, all considered to be unimpacted by pollution, were conducted to establish naturally occurring background concentrations of contaminants as a basis for establishing remediation goals for Pease (NHDES, unpublished data). Elevated concentrations of DDT compounds reflect local deposition or application probably from the 1950s and 1960s (Weston, 1994). Detailed summaries of environmental factors at each of 48 Installation Restoration Program sites have been compiled (USAF, unpublished report). On the basis of extensive assessments of sediment and water contaminant analysis and toxicity assays, remedial alternatives for sediments were evaluated (Weston, 1996). Cleanup and remediation of stream sites with contaminated sediments include Paul's and McIntyre brooks, which had elevated concentrations of pesticides, metals and PAHs of concern to ecological receptors, though not to humans (USAF, 1997). Contaminants in Lower Newfields Ditch and Flagstone Brook have been determined to pose no risk to humans or ecological receptors, and no further action has been recommended.

The Coakley Landfill is located in North Hampton 6 miles up the freshwater portion of Berry Brook. It received municipal and industrial wastes from the Portsmouth and Pease Air Force Base area between 1972-1985. In 1983, the NHDES found groundwater and surface water contamination with volatile organic compounds (VOCs) at numerous sites in the area (see Hughes and Brown, 1995). The site was added to the USEPA National Priority List in 1983, ranked number 680. The site has undergone remediation, yet VOCs are still being detected in some locations near the landfill (1993 EPA data). This became a concern to the Town of Rye and they undertook a small investigation of water quality along the whole length of Berry Brook. They sampled twice during the spring of 1995, and had samples from 9 sites along the stream, from the Coakley Landfill to the Estuary, analyzed for a wide range of contaminants (Hughes and Brown, 1995). These included 10 metals, 60 VOCs, 20 pesticides and 7 PCBs. None of the toxic organic compounds were detected in any sample. The metals were all present at low concentrations or undetectable. They found dissolved oxygen to be low near the landfill, but satisfactory at other sites. Suspended solids, dissolved inorganic nitrogen and phosphorus, and fecal indicator bacteria concentrations were all low.



Pease International Tradeport

Other Superfund sites are located within close proximity to the Great Bay Estuary. The Tolend Road site in Dover is located near the upstream portion of the Bellamy River. The Somersworth landfill is located near the Salmon Falls River.

2.3.3.3 Documented Groundwater Pollution Sources

Landfills, fuel storage, hazardous waste generators and documented groundwater pollution sources are all in GIS on the GRANIT system (Figure 2.28). A recent compilation of landfills located within the Great Bay Estuary watershed was provided by NHDES, and is presented in Table 2.5. Most of the landfills have a Groundwater Management Permit. This requires leachate monitoring, and information on flow and analytical composition are routinely submitted to NHDES for review.

TABLE 2.5 *Conditions and characteristics of active and closed landfills in the coastal region of New Hampshire.*

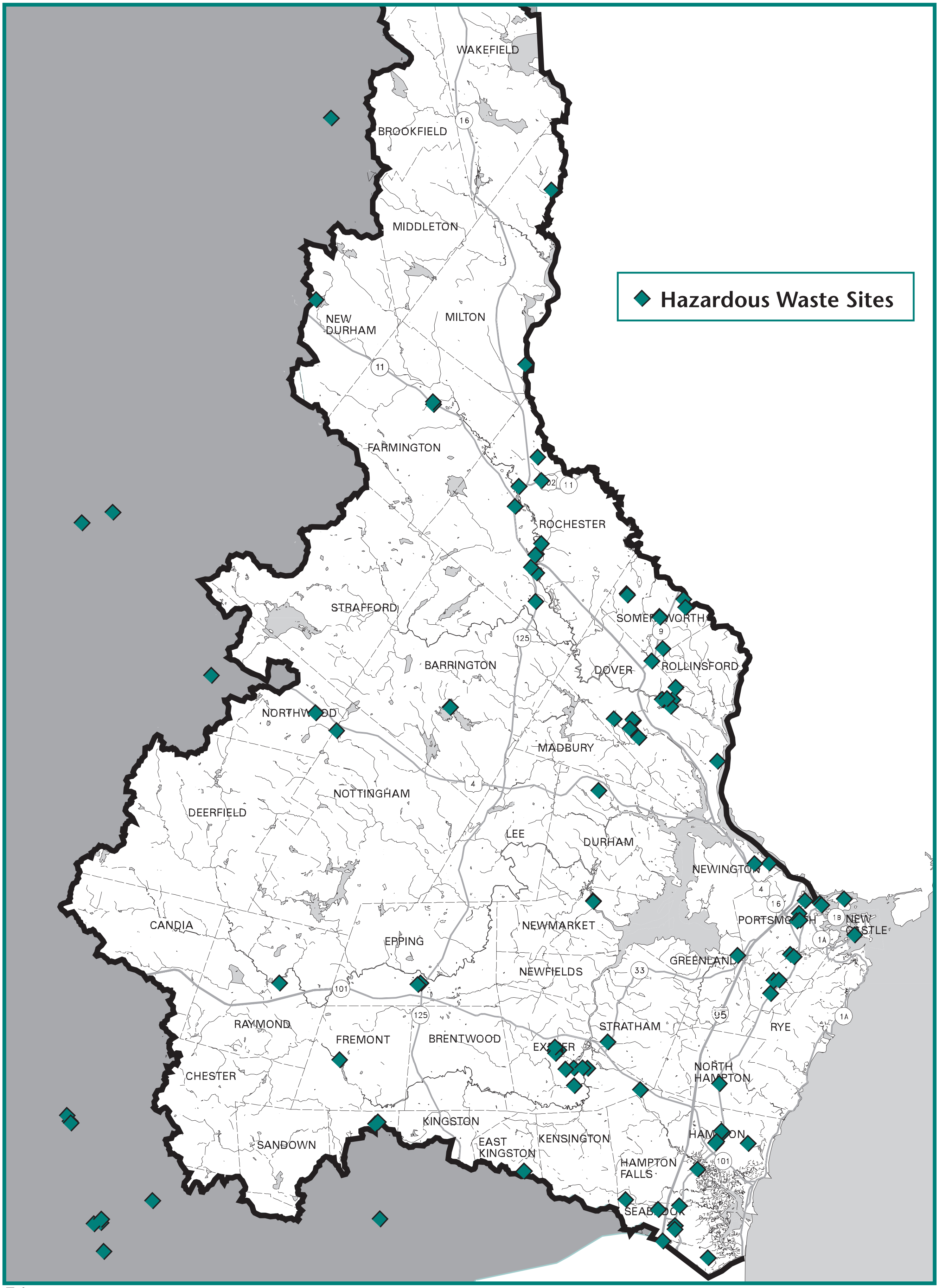
Town	Location	Start-up[1]	Active vs Closed	Lined vs Unlined	Leachate Monitored[2]	Hydraulic Connection
Barrington	Smoke St.	Early 1950s	Inactive since 1980	Unlined	Yes	
Brentwood	NO MSW 3	N/A	N/A	NIA	N/A	N/A
Brookfield	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
Candia	New Boston Rd.		Inactive	Unlined	Yes	
Chester	Route 102	Mid. 1950s	Active	Unlined	Yes	
Deerfield	Brown Rd.	1970s	Closed[4] 1996	Unlined	Yes	
Dover	Toland Road	1960	Inactive	Unlined	Yes	
Durham	Durham Pt. Rd.	1950	Inactive	Unlined	Yes	Adjacent to Horsehide Brook
East Kingston	NO MSW3 LANDFILL	N/A	N/A	N/A	N/A	N/A
Epping	Old Hedding Rd.		Inactive	Unlined	No	
Exeter	Cross Rd.	1976	Closed 1995	Unlined	Yes	
Farmington	Watson Corner Rd. (Municipal)	1940s	Active	Unlined	Yes	Water flows toward the Cocheco R.
	Watson Corner Rd. (Private)	Late 1960s	Inactive (Cardinal Landfill)	Unlined	Yes	Water flows toward the Cocheco R.
Fremont	Danville Rd.	1960s	Inactive since 1978	Unlined	Yes	Is adjacent to the Exeter R.
Greenland	Cemetery Ln.	Pre. 1900	Inactive	Unlined	No	
Hampton	Tide Mill Rd	1963	Closed 1996	Unlined	Yes	
Hampton Falls	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
Kensington	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
Kingston	Route 125	1920s	Active	Unlined	Yes	
Lee	Mast Rd.		Inactive	Unlined		
Madbury (Madbury Metals)	Route 155	Late 1970s	Closed[1] 1995	Unlined	Yes	
Middleton	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
New Castle	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
New Durham	Old Rte 11	Early 1970s	Inactive	Unlined	No	

Conditions and characteristics of active and closed landfills in the coastal region of New Hampshire (continued).

Town	Location	Start-up[1]	Active vs Closed	Lined vs Unlined	Leachate Monitored[2]	Hydraulic Connection
Newfield	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
Newington	Pease Tradeport	Mid. 1950s	closed[6] 1996	Unlined	Yes	
Newmarket	Ash Swamp Rd	1950	Closed 1995	Unlined	Yes	
Northwood	Route 4		Inactive	Unlined	No	
North Hampton	Coakly Superfund Site	1972	Inactive closure expected 1997	Unlined	Yes	
Nottingham	Freeman Hall Rd	1973	(Ash Pile] Active	Unlined	No	
	Freeman	1960s	Active[8]	Unlined	Yes	
Portsmouth	Mirona Rd	1950s	Inactive	Unlined	No	
	Jones Ave. Ash LF	1940s	Closed 1991	Unlined	Yes	
	PSNH Schiller Sta Woodbury Ave		Closed 1980s	Unlined	Yes	
Raymond	Prescott Rd.		Closed	Unlined	Yes	
Rochester	Turnkey LF	1980s	Active	Double Lined	Yes	
	Old Dover Rd	Closed	1980s	Unlined	Yes	
Rollinsford	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
Rye	Breakfast Hill Rd		Closed 1988	Unlined	Yes	
	Grove Rd		Inactive	Unlined	Yes	
Sandown	NO MSW LANDFILL	N/A	N/A	N/A	N/A	N/A
Seabrook	Rocks Rd.		Inactive	Unlined	No	
Sommersworth	Blackwater Rd.	1930s	Inactive Superfund Site	Unlined	Yes	
Strafford	Nelson Rd.		Inactive	Unlined	No	
Stratham	Union Rd.	1950s	Closed 1995	Unlined	Yes	
Wakefield	Route 153	1974	Active	Unlined	Yes	

1. A blank box indicates there is insufficient information on file to determine the date the landfill began accepting waste.
2. Leachate is monitored by the use of groundwater monitoring wells and surface water stations a: the landfill site.
3. MSW = Municipal Solid Waste.
4. Closed = Closed in accordance with State approved test" plans.
S. The Madbury Metals landfill c contains automobile shredder residue.
6. There were a total of five MSW, three Construction/Rubble Dump landfills and one paint can disposal area at the former Pease Air force Base. Four MSW landfills were combined and closed as one site, while the fifth is a stump disposal area which is inactive. Two of the Rubble Dumps and the Paint can area continue to be monitored.
7. A file review proved inconclusive on whether PSNH had received state approval for the landfill closure design.
8. The landfill in Nottingham is ~ construction and demolition debris landfill.

FIGURE 2.28 *Hazardous waste sites and landfills in the coastal region of New Hampshire.*



2.3.3.4 Oil Spills

There have been many oil spills of a wide range of volumes in coastal New Hampshire. During 1975-79 there were 103 oil spills in public waters in the 17 coastal communities (SRRC, 1981). The most significant spills included the tanker *Athenian Star* (10,000 gallons of diesel fuel) in 1975, Bouchard Barge #105 (8000 gal. #6 fuel oil) in 1978 and the tanker *New Concord* (25,000 gal. #6 fuel oil) in 1979, mostly associated with the oil terminals in Portsmouth and Newington on the Piscataqua River. Even though smaller spills were more frequent (94), nine spills of >500 gallons constituted 95.3% of the spilled oil. The impacts of the oil spills included fouling of beaches, shorelines, boats, docks, fishing gear and lobster traps. Many people reported that the shellfish beds in front of their houses were destroyed and that the marsh grass along the shoreline was removed because it trapped and retained oil. Many claims filed by lobstermen and shoreline residents were still pending a year and a half after some spills.

A 1981 NHF&G study (Nelson, 1982) was done specifically to serve as a baseline for assessing future oil spill impacts to estuarine resources. As a means of assessing the impact of oil spills on sediments, sediments were collected monthly at 24 intertidal and subtidal sites throughout the Great Bay Estuary and analyzed for hydrocarbons. Nelson (1982) reported the results of analyses for PAHs and alkanes for February, 1981 at both intertidal and subtidal sites at eight different stations. Concentrations were reported for 13 different PAHs, ranging from 0 for numerous PAHs to >1000 ng/g sediment for chrysene and benzo[a]anthracene at Nobles I., Cedar Pt., Royall's Cove and Fox Pt. Alkane analysis was reported as concentrations for even and odd-numbered carbons in chains ranging from 14 to 32 carbons. Total alkane concentrations ranged from 707 ng/g sediment to 24,960 ng/g sediment. Sites with the highest concentrations included Rollins Farm (>14,800 ng/g), Broad Cove (>17,000 ng/g) Royalls Cove (>24,900 ng/g) in either intertidal or subtidal sites. Evidence of contamination from oil spills was evident at all sites, suggesting that oil spilled mainly in the lower estuary was likely transported to the upper estuary.

At the present time, NHDES keeps records of all oil spills, including those that are spilled into surface waters. NHDES also has an oil spill clean up program. The NH Coastal Program keeps records of oil spills in the communities included on the coastal program.

The most recent significant oil spill in the coast of New Hampshire occurred in the Piscataqua River on July 1, 1996. It involved a spill of ~1,000 gallons of #6 fuel oil from the vessel Provence. The various types of compounds in the oil had different dispersion behavior, with some oil sinking and other fractions floating. The floating oil was collected along the shoreline of Little Bay, and the portion that sank is probably now associated with Little Bay sediments. Much of the oil sank in Little Bay, and the impact to biota was under investigated (NHF&G, 1996). Chase et al. (1997; 1998) reported elevated concentrations of PAHs in blue mussels at Dover Point 16 days after the spill in comparison to 1994 concentrations (Chase et al., 1996a). Low molecular weight PAHs decreased in concentration or disappeared in samples collected three and fifteen months after the spill, but concentrations of high molecular weight (> 5 rings) PAHs persisted and were still significantly higher than in 1994 tissue. Samples of both blue mussels and oysters from Fox Point collected 16 days after the spill had concentrations of PAHs approximately twice as high as seen at Dover Point. This difference is probably a function of where the oil was eventually deposited after initial transport via water currents soon after the spill.

In 1998, the NHDES joined efforts with the Gulfwatch program through UNH/JEL to expand the use of monitoring blue mussel tissue for toxic contaminants in New Hampshire waters (Jones and Landry, 2000). One key goal is to establish a baseline of data that could be used to monitor recovery in the event of a future oil spill. New monitoring sites have been established that bracket the major oil storage and off-loading facilities on the Piscataqua River and in other areas of the estuary that could be impacted by spills.

2.3.3.5 Fertilizer and Pesticide Applications

Historically, agricultural activities are associated with significant fertilizer and pesticide applications. The small number and sizes of crop-producing farms in coastal New Hampshire make agriculture less significant, and the contributions of golf courses and residential lawns has become relatively more significant. Use of all types of pesticides in Rockingham and Strafford counties has increased since 1965 (NHCRP, 1997). In 1994, 281,706 lbs of >250 pesticides were used in NH, with 1,000 to 10,000 lbs/y in estuarine drainage areas.

There are at least ten golf courses in the coastal communities of New Hampshire. Many are inland, but a few are in close proximity to estuarine surface waters. All golf courses need to use fertilizers and pesiticides to maintain the high quality turf on fairways and greens. Pesticides transported to estuaries via runoff or groundwater can cause harm to non-target estuarine organisms. Pesticide use at NH golf courses is regulated through a New Hampshire Pesticide Board (Department of Agriculture) permitting process. A survey of groundwater samples from 25 shallow wells at agricultural sites and golf courses, some of which were in the coastal area, showed no detectable pesticides, and metal concentrations were all within drinking water standards (NHDHHS, 1986).

Runoff and groundwater can also contain nutrients from fertilizers that may contribute to nutrient overenrichment. A drainage swale downgradient from the Rockingham Country Club in Newmarket had the highest loading rate for nitrate (~2.7 kg nitrate/d during high flow) than any other tributary to the Squamscott River (Jones and Langan, 1995c). Possible upstream sources were investigated and no significant source other than the golf course was apparent.

The Wentworth-by-the-Sea golf course uses a number of strategies to manage fertilizer and pesticide applications and minimize environmental impact because they use both on land that is immediately adjacent to Little Harbor (Rye-Wentworth Impact Assessment Report, 1990). A slow-release fertilizer (24-4-12) is applied to fairways, tees and greens in May, June and September at annual rates ranging from 130-218 lbs/acre of nitrogen and 22-36 lbs/acre phosphorus. Roughs are not fertilized. Grass clippings are returned directly (mulched) onto fairways. Tee and green clippings are collected and spread on the roughs. Water sample analysis suggested that the fertilizers applied at the course have little impact on the water quality of the harbor (Jones and Langan, 1995c). Insecticides are not used routinely or on a large scale. Instead, an integrated pest management system is employed and pesticide application is limited to spot application to control grub infestation. Preventative treatment for snow mold fungus is applied only to tees and greens. Heavy metal (mercury) based compounds are not used. All materials are applied conservatively with particular caution paid to adjacent surface waters and wetland buffer zones. Equipment used for applications is field-rinsed, and the diluted rinse water is sprayed onto the fairways to prevent a large volume of this water being washed into maintenance facility storm drains (Rye-Wentworth Impact Assessment Report, 1990).

Some other golf courses are in relatively close proximity to estuarine waters and tributaries. Portsmouth Country Club is located in Greenland on the southeastern shore of Great Bay, the Rochester, Farmington and Cocheco country clubs are near the Cocheco River, the Exeter Country Club is near the Squamscott River, and Pease Golf Course is near the shores of Great Bay.

Within salt marshes, human nuisances such as mosquitos and green-head flies are managed by seacoast towns that collectively spend approximately $100,000 each year (USDA 1994); ironically, most of the effort to control these pests occurs in degraded marshes (see habitat loss section). The NH Division of Pesticide Control has provided information on the coastal towns involved and the major contractors. The towns include Newcastle, Newfields, Stratham, Hampton Falls, Portsmouth, Hampton, Rye, Newmarket, Exeter, Newington, Seabrook and the Great Bay National Refuge. The towns conduct integrated systems of control, using both adulticiding and larviciding techniques. Insecticides used include GB-111 and VectoBac 12AS, CG and G. The larvicidal insecticides used typically depend on the activity of the bacterium *Bacillus thuringiensis* var. *israelensis*, and the adulticides are often pyrethroids.Organophosphate insecticides are also used.

2.3.3.6 Atmospheric Deposition

In an effort to refine and regionally focus the issue of atmospheric deposition of mercury, representatives of the regions state air, water, waste and public health divisions and Environment Canada formed a Mercury Workshop. This group recently published their findings (NESCAUM, 1998). The Workshop concluded that about 47% of mercury deposition in the region originated from sources within the region, 30% from U.S. sources outside the region, and 23% from the global atmospheric reservoir. This report has provided the impetus for a concerted regional effort to reduce mercury emissions. On June 8, 1998, the New England governors and eastern Canadian premiers agreed to cut regional mercury emissions from power plants, incinerators, and other sources in half by the year 2003 (Boston Globe -6/9/98).

The USEPA has monitored 70 toxic volatile compounds, including 56 volatile organic compounds (VOC) at Portsmouth and three other sites statewide since 1989 (NHCRP, 1997). Anthropogenic sources of VOCs include industrial processes, solvents, oil-based paints and automobiles. In 1994, the volume decreased to 23,174,000 tons, down from 30,646,000 tons in 1970. Most of the reduction came from automobiles, as the amount decreased from 12,972,000 to 6,295,000 from vehicles. Of the 70 compounds monitored, 37 have disappeared since 1987, and 15 have decreased in concentration.

A summary of recent existing input and output data for four inorganic and nine organic contaminants in the Gulf of Maine identified major data gaps in the current understanding of atmospheric deposition of contaminants (McAdie, 1994). Numerous papers were presented at a recent conference on regional atmospheric Hg deposition (EMAN, 1996). Gaseous mercury concentrations in the atmosphere over the Gulf of Maine were reported to range from 0.4 to 2.0 ng/m3. The concentrations generally vary inversely with altitude. Municipal and medical waste incineration is probably a significant localized (30-50 mile radius) source of Hg deposition in New Hampshire. In Maine, measurements of mercury in rain and snow showed ranges of 5-15 ng/L, giving wet deposition values of about 6-10 µg/m2/y. A new atmospheric monitoring station has been established at Newcastle, NH. Data collected are providing information on atmospheric mercury deposition in the coastal New Hampshire area as part of the national Mercury Deposition Network (MDN). Comparison with an inland MDN site at Laconia, NH, suggested that New Castle may be receiving greater mercury deposition than inland areas, along with other coastal sites in new England (VanArsdale et al., 1998).

2.3.3.7 Summary

Aside from historically resuspended contaminated sediments, the most significant documented sources of contaminants are stormwater runoff, oil spills and Superfund sites located adjacent to the Great Bay Estuary. All three source categories are receiving attention by state, federal and private agencies to mitigate contamination in the remaining source areas of New Hampshire. For some contaminants like mercury, atmospheric deposition is suspected to be a significant source, but is at present not well documented. Continued reductions of external sources of contaminants is important because of the existence of elevated contaminant concentrations from historical sources in some areas.

2.3.4 CONTAMINANT AND HYDRODYNAMIC MODELING

Mathematical computer modeling of circulation and tidal flow in the Great Bay Estuary was first done in the 1970s (Celikkol and Reichard, 1976; Brown and Arellano, 1979). The early two dimensional model examined the movement of water up the main stem of the Estuary and calculated the flushing time and tidal exchange for the various parts of the estuarine system (Swift and Brown, 1983; Short, 1992b). More detailed two dimensional models have been developed to examine the path that oil might take if a spill were to occur in the Estuary (Swift and Celikkol, 1983). The primary focus of the oil spill model was on the Piscataqua River near the oil loading terminals. The model included the upper Estuary, but it was never calibrated for Great Bay proper.

Recent efforts have begun to model the hydrodynamics and current flow patterns in Great Bay proper as part of an effort to develop modeling capabilities for simulating hydrodynamic flows in estuaries having intertidal areas (Ip et al., 1997). This model provided the first detailed hydrodynamic assessments for Great Bay and successfully simulated the movement of water on and off the extensive intertidal mudflats within that system. This two dimensional finite element model for Great Bay, currently under development at Dartmouth College, produces fine scale output of current velocities and tidal variations within Great Bay and upper Little Bay. The problems of model simulation within intertidal estuaries have been resolved, but the Great Bay model has not yet been field verified.

A finite element, two dimensional hydrodynamic model has been adapted to the entire Great Bay Estuarine system as part of the US Navy Ecological Risk Assessment Study (Pavlos, 1994). The WASP4 model, originally developed by the EPA, was used to estimate the distribution of lead throughout the Great Bay Estuary, assuming discharges were occurring at the Portsmouth Naval Shipyard (Chadwick, 1993; Pavlos, 1994). The model includes the simulation of dissolved substances within the water column throughout the lower portions of the Estuary (TOXIWASP, Pavlos, 1994). The TOXIWASP model was used to examine salinity distribution as well. The development of an improved version of the WASP model and the need for better accuracy in model predictions lead to the application of the WASP5 model to the Great Bay Estuary and a series of simulations, again looking at the transport of lead from sources around the shipyard as well as sources elsewhere in the Estuary (Scott, 1997). The focus of the WASP5 model was the Piscataqua River and Portsmouth Harbor although it was fit to the entire Estuary. This model was successful in predicting the transport of lead throughout the lower part of the Estuary and in determining sites where elevated concentrations of lead might accumulate.

WASP has recently been used to model nonpoint source pollution in the tidal portion of the Oyster River (Swift et al., 1996). Different programs within WASP were used to model currents and water levels, salinity, bacteria, nutrients and dissolved oxygen. The model exercise found that the flushing time of the river is 3 days. The model was also used to simulate contaminant distributions for an effluent release from the Durham WWTF, a significant rainfall event, and for average conditions. The results were relatively effective for simulating trends and processes when compared to field data collected as part of two previous studies (Jones and Langan, 1993a, 1994c).

WASP was also used by the State of Maine (Mitnick, 1994) to determine the reduction in phosphorus from WWTF required to meet the strict Maine WQCs for chlorophyll in the freshwater portions of the Salmon Falls River. The major WWTF included were at Berwick, ME and Somersworth, NH. The results suggested drastic reductions in phosphorus discharges would be needed. Experimental reductions in phosphorus at the WWTF confirmed that reductions in chlorophyll in the freshwater portion of the river were possible (Mitnick, 1994).

2.3.5 PUBLIC HEALTH RISKS AND ECOLOGICAL IMPACTS

New Hampshire coastal waters are popular areas for commercial and recreational fishing and recreational shellfishing. In addition, the area is noted and valued for its relatively pristine conditions, and the ecological integrity of the coast is an important resource. One threat to both public health and ecosystem integrity is the presence of toxic contaminants. The NHDHHS and other state agencies monitor contaminants and assess the risks to humans. They provide direct access to consumption advisory information via 1-800-852-3345 ext. 4664. At present, there are advisories based on elevated Hg in inland lakes and rivers, and two advisories in New Hampshire related to consumption of marine fish, both based on elevated PCBs (Table 2.6; NHDES, 1996b). These advisories are based on three studies conducted more than nine years ago. One of the first studies for shellfish from coastal New Hampshire was by Isaza et al. (1989). The results suggested that lead, PCB and PAH concentrations were elevated and warranted further study. To further determine how shellfish may impact human health, another study was conducted by NHDHHS (Scwalbe and Juchatz, 1991). As a result of the PCB concentrations found in lobster tomalley in their study, DHHS issued a consumption advisory for lobster tomalley in the Great Bay Estuary. There was also an advisory for consumption of coastal bluefish in New Hampshire issued in 1987 because of elevated PCB concentrations found in bluefish from sites along the Atlantic Coast (NOAA, 1987). These advisories are thus based on small, relatively old databases. More recent studies have provided newer and more comprehensive information on tissue body burdens of

Recommended consumption advisories for fish from the New Hampshire Department of Health and Human Services. From NHDES (1996b).

TABLE 2.6

	Who We're Concerned About	Species of Concern	Recommendations
General Advisory For All Inland Freshwater Bodies	• Women of reproductive age	All species	Limit to one 8-oz. meal per month
	• Children 6 years of age or younger	All species	Limit to one 3-oz meal per month
	• All other consumers	All species	Limit to four 8-oz meals per month
Androscoggin River (from Berlin to the Maine border)	• Pregnant and nursing women	All species	Avoid consumption
	• All other consumers	All species	Limit to one or two 8-oz. meals/year
Great Bay Estuary	• Pregnant and nursing women	Lobster Bluefish	Limit consumption; avoid tomalley Avoid consumption
	• Children under 15	Lobster Bluefish	Limit consumption of tomalley Avoid consumption
	• All other consumers	Lobster Bluefish	Limit consumption of tomalley Avoid fish over 20 in. or 4 lbs; prepare according to guidelines
Connecticut River	• All consumers	All species	Prepare according to guidelines
Horseshoe Pond	• All consumers	Largemouth Bass	Avoid consumption

contaminants for a variety of animal and plant species.

Contaminant concentrations in blue mussels, other shellfish, lobsters, winter flounder and marine plants have been reviewed and summarized. The database available for blue mussels (*Mytilus edulis*) is the largest of any organism, with up to 85 sample analyses for each contaminant (Table 2.7. A more detailed summary is presented in Appendix H. Blue mussels are commonly used as an indicator for habitat exposure to organic and inorganic contaminants. Bivalves such as *M. edulis* have been successfully used as indicator organisms in environmental monitoring programs throughout the world (NAS, 1980; NOAA, 1991; Widdows and Donkin, 1992; O'Connor, 1992; O'Connor and Beliaeff, 1995; Widdows et al., 1995; Jones et al., 1998) to identify variation in chemical contaminants among sites and contribute to the understanding of trends in coastal contamination.

Blue mussels are a useful indicator organism for the following reasons: they are abundant within and across coastal New Hampshire; they are easy and inexpensive to collect and process; much is known about mussel biology and physiology; mussels are a commercially important food source (although in New Hampshire there is only recreational harvesting of mussels) and therefore a measurement of the extent of chemical contamination is of public health concern; adult mussels are sedentary, thereby eliminating the complications of interpreting results introduced by mobile species; mussels are suspension-feeders that pump large volumes of water and concentrate many chemicals in their tissues making it easier to detect trace contaminants; and the measurement of chemicals in bivalve tissue provides an assessment of biologically available contamination that is not always apparent from measurement of contamination in abiotic environmental compartments (water, sediment, and suspended particles). They also have well-defined limitations. One limitation is that they are only mildly tolerant of low salinities, and alternative shellfish (oysters, clams) may be required for areas such as Great Bay and some tributaries where salinities can be too low.

A summary of the data for mussels in coastal New Hampshire and nearby areas in Maine and Massachusetts is presented in Table 2.7. More detailed presentation of specific organic contaminants is available in Appendix H and in the reports that served as sources of this information. A series of "Guidance Documents" have recently been published by the USFDA (1993) for cadmium, chromium, lead and nickel "alert" levels. The levels do not warrant issuance of health advisories, but serve as useful target concentrations for assessing potential health risks from seafood consumption. The data in Table 2.7 show no metal other than lead came close to the alert levels. Lead concentrations in mussels exceeded the guideline level of 11.5 µg/g dry weight in nine samples at five sites around Seavey Island in Portsmouth Harbor and at one site in the Lamprey River. The highest concentration was 76 µg/g at Henderson Point on the southern end of Seavey Island. The other sites with concentrations >11.5 µg/g had values of 12.0-32.4 µg/g.

In 1997, mussels from Rye Harbor, Dover Point and Clarks Cove on Seavey Island had greater tissue Hg levels (>0.64 µg/g) than any of the other 22 sites monitored (Chase et al., 1998). An analysis of the Gulfwatch data from 1995 showed that the highest concentrations of cadmium and chromium from amongst the 14 sites monitored throughout the Gulf of Maine were found in mussels from Dover Point (Chase et al., 1996). For the first five years, 1991-1995, samples from Shapleigh I., Dover Point and Clark Cove had the 2nd, 4th and 7th highest chromium concentrations in the Gulf of Maine from amongst 59 sites (Jones et al., 1998). Samples from the same three sites and Little Harbor had amongst the top ten concentrations in the Gulf of Maine for lead, mercury, nickel, zinc, aluminum and iron, while the 1995 Dover Point sample with a high cadmium concentration was the highest in the Gulf for the five year period.

TABLE 2.7

Toxic contaminant concentrations in bivalve shellfish tissue from sites in Coastal New Hampshire and in Maine sites in Portsmouth Harbor: 1985-1997.

	USFDA Action Level for shellfish	Blue mussels *Mytilus edulis* Tissue Concentrations Average	Range	No. of samples	American oyster *Crassostrea virginica* Tissue Concentrations Average	Range	No. of samples	Soft shell clam *Mya arenaria* Tissue Concentrations Average	Range	No. of samples
Trace metals		µg/g*	µg/g		µg/g*	µg/g		µg/g*	µg/g	
Ag		0.5	0.03 to 2.8	66	17.0	12.3 to 22.6	5			0
Al		282	77 to 650	40						
As		8.5	5.1 to 13.5	36	6.5	4.1 to 10.1	13	20.6	20.6	1
Cd	25	2.3	0.1 to 9.3	85	4.5	3.5 to 6.8	5	1.0	0.3 to 1.4	8
Cr	87	5.1	1.5 to 57	85	2.7	1 to 4.5	15	11.1	4.3 to 26.7	8
Cu		9.6	5.5 to 45.5	83	215	114 to 301	7	13.3	11 to 15	2
Fe		572	209 to 1,300	46						
Hg	6.7	0.47	0.13 to 1.1	73	0.61	0.07 to 1.1	13	0.35	<0.2 to 0.42	9
Ni	533	2.6	1.1 to 16.7	72	3.2	2.7 to 4.1	5	9.3	9.3	1
Pb	11.5	8.4	1.9 to 76	85	2.2	0.61 to 5.2	17	13.1	5.6 to 36	9
Zn		122	80 to 270	85	5383	3,770 to 6,000	7	70	59 to 80	2
Toxic Organics			ng/g	ng/g		ng/g	ng/g		ng/g	ng/g
PCBs	13000	339	5 to 2,540	42	199	189 to 246	6	161	<67 to 247	8
PAHs		3831	69 to 73,300	42	628	442 to 1145	8	26,013	<0.67 to 38,000	7
Cl'd pesticides	33000	20	3.5 to 51.8	24	105	88.4 to 159	6			0
Dioxins, Furans, Planar CBs										
CA tolerance level=133pg/g†		pg/g	pg/g							
CB/PCDD/ PCDF TEQ††		8.27	1.70 to 17.5	4						

* Dry tissue weight. To convert original data expressed as wet weights, assume 12% (oysters), 15% (mussels) and 16% (clams) dry weight.

† CA tolerance level (133 pg/g): Health Canada tolerance level for seafood consumption for 2,3,7,8-TCDD (133 pg/g DW = 20 pg/g WW; assume 15% solids).

†† Toxic Equivalency Concentrations for planar chlorinated biphenyls (CBs), dibenzo-dioxins (PCDD) and dibenzo-furans (PCDF) are based onn standardized factors for determining additive relative toxicities of these compounds that share a similar mode of toxicity.

Concentrations of organic contaminants in mussels in Table 2.7 are compared to FDA Action Levels for fish and shellfish. The organic contaminants analyzed that have Action Levels included PCBs, dieldrin, aldrin, chlordane, heptachlor, heptachlor epoxide, DDT and methyl mercury. Action Levels for total PCB and DDx are presented in Table 2.7. All reported organic concentrations are less than, and in most cases, far below the action levels. However, the PCB concentrations at the Dry Docks on Seavey Island and at sites in the upper Piscataqua River were only 5-8 times lower than the action limit of 13 µg/g.

The effects of contaminants on the physiology of mussels has also been assessed in a few studies. Gilfillan et al. (1985) found effects of contaminants on mussel physiology assays were more related to metals than to aliphatic or aromatic hydrocarbons in Portsmouth Harbor. They found Cd, Zn, Ag, Cr and Cu affected activities of glucose-6-phosphate dehydrogenase, aspartate amino transferase and scope for growth assays in mussels for some sites some of the time, although effects were not consistently measured at any specific site. Jones et al. (1998), reported that copper and zinc concentrations in mussel tissue from Little Harbor and Shapleigh Island in 1991 and 1992 exceeded critical body residue levels, or the lowest concentrations at which observed toxicity effects have been observed. Gulfwatch and Portsmouth Naval Shipyard studies have also reported extensive information on mussel growth and condition index, as well as limited information on scope for growth of mussels. The condition index data for indigenous and deployed mussels in New Hampshire indicate mussel growth and physiological condition are within normal ranges, although somewhat lower than other areas of the Gulf of Maine (Chase et al., 1997; 1998; Jones et al., 1998). The scope for growth measured in deployed (caged) mussels in Cutts Cove was the only indication of stress in deployed mussels in Portsmouth Harbor (NCCOSC, 1997).

A recent report from the USEPA (Metcalf and Eddy, 1995) reviewed published contaminant databases and determined background concentrations for contaminants in shellfish in New England and the North Atlantic continental shelf areas. Comparison of the lowest observed contaminant concentrations in New Hampshire mussels to the regional background concentrations showed concentrations of cadmium, PAHs, PCBs and DDx were close to background concentrations at some New Hampshire sites (Table 2.8). Other contaminants, especially arsenic, mercury and zinc, were present only at much higher concentrations, suggesting ubiquitous, regional sources of these contaminants.

Other studies have reported contaminant concentrations in different shellfish species. These data are summarized in Tables 2.7 and 2.9, and in greater detail in Appendix H. Isaza (1989) also analyzed clams (*Mya arenaria*), lobsters and sediments. Nelson (1986) analyzed oysters from four sites in the Great Bay Estuary for chromium and lead. Oysters were analyzed for a range of contaminants as part of the Portsmouth Naval Shipyard study (Johnston et al., 1994; NCCOSC, 1997). Langan and Jones (1995c) analyzed oyster (*Crassostrea virginica*) samples from Great Bay, and compared results to previous studies. Comparison of concentrations to USFDA Action Levels shows only lead in the clams from Hilton State Park at Dover Point exceeded the 11.5 µg/g Action Level. Relatively high concentrations of mercury in oysters, PAHs in clams and chromium in clams were also observed (Table 2.7). The lowest DDx concentrations in oysters were relatively close to background concentrations while concentrations of cadmium, chromium and PCBs were relatively high. Conversely, most contaminants that could be compared showed relatively low, and sometime lower, concentrations compared to background concentrations.

Numerous studies have reported contaminant concentrations in different types of lobster tissue (Table 2.9). PCB

Published background concentrations in New England waters (Metcalf and Eddy, 1995) and observed lowest concentrations for contaminants in blue mussels from coastal New Hampshire and Portsmouth Harbor.

TABLE 2.8

	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn	PAHs total	PCBs total	DDT and metabolites
Background concentrations* (Gulf of Maine)	0.23	0.20	0.30	1.40	0.01	0.30	0.60	3.70	0.04	0.01	0.01
Lowest concentrations† (New Hampshire)	5.10	0.10	1.50	5.50	0.13	1.30	2.10	80	0.07	0.01	0.01
USFDA Action Levels		25	87		6.7	533	11.5			13	33

* Background concentrations of contaminants in shellfish in New England and North Atlantic continental shelf area. From Metcalf and Eddy (1995).

† Lowest (background) concentrations of contaminants in shellfish in New Hampshire/Portsmouth Harbor.

concentrations in adult muscle and viscera tissue from Pierces Island in Portsmouth Harbor were in excess of the 13 μg/g action limit. These data are from the initial study that served as the basis for the lobster consumption advisory in New Hampshire (Isaza et al., 1989). Relatively high concentrations of cadmium and mercury were also observed in some different lobster tissue from various areas around Portsmouth Harbor.

Plant tissue levels of contaminants have also been reported (Table 2.10). As part of the Portsmouth Naval Shipyard study (Johnston et al., 1994), contaminants were measured in eelgrass (*Zostera marina*), fucoid algae (*Ascophyllum nodosum*) and winter flounder (*Pleuronectes americanus*). In the winter flounder samples, contaminant concentrations were well below FDA action levels. Concentrations of metals in eelgrass and fucoid algae showed elevated concentrations of some metals, and apparently different accumulation rates for some metals compared to mussels. Fish tissue from Peverly Ponds and Bass Pond at Pease AFB indicated all organic contaminants were below detection limits, except for DDT compounds (NHDES, unpublished data).

Sowles et al. (1996) reported heavy metal and organic contaminant concentrations in small mouth bass and white suckers from the Salmon Falls River. Mercury concentrations were similar to concentrations found in fish from lakes and ponds that prompted a fish consumption advisory in Maine. PCB and DDT concentrations also exceeded some human health threshold levels, and both metal and organic contaminant concentrations at some sites were near concentrations considered harmful to wildlife.

There have been numerous studies on contaminant concentrations and impacts on birds in the Gulf of Maine region. In addition, NHDES contracted in 1997 with a private company to provide wildlife rescue and rehabilitation in response to oil spills.

In general, only rare occurrences of tissue contaminant concentrations exceeded USFDA Action Levels. However, USFDA Action Levels may be higher than concentrations that can cause human and wildlife health problems. The relatively high concentrations for several trace metals and toxic organic contaminants are a concern, especially when they are consistently well above regional background concentrations. The cumulative effects of elevated concentrations of multiple contaminants are not well characterized, but certainly present a problem for the living resources and humans that inhabit the coastal areas of New Hampshire. Recent studies on the role of

TABLE 2.9 *Toxic contaminant concentrations (dry weight) in lobsters and winter flounder tissue from sites in New Hampshire, Portsmouth Harbor and the Isles of Shoals: 1985-1997.*

		Lobster *Homarus americanus*			Winter flounder *Pleuronectes americanus*		
Contaminant	USFDA Action Level	Tissue Concentration Average*	Range	# of samples	Tissue Concentration Average	Range	# of samples
Trace metals	μg/g	μg/g	μg/g		μg/g	μg/g	
Ag		1.0	0.25 to 3.01	24	0.3	0.008 to 0.66	4
As		13	4.35 to 19.7	24	4.4	2.10 to 6.41	4
Cd	25	4.7	0 to 15.4	27	0.1	0.01 to 0.16	4
Cr	87	0.4	0.12 to 1.6	28	0.4	0.23 to 0.73	4
Cu		112.3	15.3 to 332	25	10.3	0.27 to 22	4
Hg	6.7	0.6	<0.14 to 2.39	26	0.15	0.10 to 0.21	3
methyl Hg	6.7	1	0.07 to 4.61	11	0.15	0.05 to 0.25	2
Ni	533	0.67	0.41 to 1.81	27	0.49	0.18 to 0.65	4
Pb	11.5	0.2	0.04 to 0.41	28	0.2	0.06 to 0.37	4
Zn		95.3	58.5 to 147	28	64.6	16.4 to 114	4
Toxic organics	ng/g	ng/g	ng/g		ng/g	ng/g	
PCBs	13000	1561	11.3 to 66,400	27	281	51.5 to 938	4
PAHs		588	47.2 to 87,600	24	479	17.2 to 531	4
Cl'd pesticides	33000	269	2.01 to 791	28	97	6.61 to 192	4

* Lobster tissue includes samples of tail, claw, hepatopancreas, viscera, cooked meat, cooked tomalley, for adults and juvenile animals.

TABLE 2.10 *Trace metal contaminant concentrations (μg/g dry weight) in marine plant tissue at sites in Portsmouth Harbor and Great Bay Estuary. Data from NCCOSC, 1997.*

Trace metal	*Zostera marina* leaves	*Zostera marina* roots	*Spartina alterniflora*	*Spartina patens*	*Ascophyllum nodosum*
Ag	0.68	0.66	0.22	0.14	0.49
As	1.3	4.5	1.2	1.2	15.2
Cd	1.25	0.53	0.07	0.10	0.55
Cr	1.7	9.2	2.0	2.3	0.73
Cu	15.5	16.9	2.1	2.8	16.9
Hg	0.02	0.05	0.01	0.02	0.04
Ni	1.82	3.09	0.69	0.98	1.83
Pb	2.4	10.9	0.97	1.8	2.3
Zn	72	57	31	27	78

*From NCCOSC, 1997.

many of the same contaminants as endocrine disruptors, especially during critical early life stages of biota, is cause for concern for very low contaminant concentrations. Continued assessments of contaminants in biota, like the Gulfwatch program, are important tools for assessing potential risks and determining trends in contaminant distribution and fate. More studies of biological effects would be useful to determine the overall toxicity of contaminants in the environment in the more contaminated estuarine areas. The detection of contaminants in New Hampshire shellfish that are close to background concentrations suggests that sites where these same contaminants are present at elevated concentrations may indicate localized sources.

INORGANIC AND ORGANIC NUTRIENTS

Eutrophication of estuarine and coastal waters resulting from excess nutrient input from anthropogenic sources has emerged as a significant problem for many coastal areas. The two most important nutrients in terms of pollution are nitrogen and phosphorus, since they are most commonly the limiting nutrients in aquatic ecosystems, though carbon, silica and trace metals such as copper and iron also play a role in primary productivity. In marine and estuarine waters, nitrogen is generally believed to be the primary limiting nutrient, though phosphorus has been identified as the limiting factor in some systems. In addition to the concentrations of nitrogen and phosphorus, the N:P ratio may also be important for some species of algae.

The biological effects of nutrient enrichment can range from subtle to extreme. Species shifts in phytoplankton communities can result in unfavorable conditions for estuarine biota, particularly for filter feeders such as bivalve molluscs. Massive blooms of phytoplankton can reduce water clarity, shade submerged aquatic vegetation (SAV), and reduce water column oxygen concentration due to nighttime plant respiration and oxygen consumption. Blooms of nuisance macroalgae can replace more desirable forms of vegetation and create hypoxic or anoxic conditions that can impact fish and invertebrates. Conditions resulting from nutrient enrichment can affect recreational activities such as fishing, boating and swimming as eutrophic systems can be most unappealing for these activities. Nutrient enrichment is also suspected to be a factor in blooms of harmful, toxin-producing algae in coastal and offshore waters. Finally, sources of biodegradable organic nutrients can be a direct cause of hypoxia and anoxia as heterotrophic bacteria can rapidly consume dissolved oxygen as they decompose organic substrates.

Assessing the trophic status or the degree of nutrient enrichment of any water body necessitates the measurement of a suite of parameters, since no single measurement can clearly depict trophic status (Kelly, 1991). In addition, the geometry (depth, width, length) and flushing characteristics or residence time of water masses are important factors in determining the susceptibility of any water body to eutrophication (Kelly, 1997). Measurements of dissolved nitrogen and phosphorus (inorganic and organic), turbidity or suspended solids, particulate organic matter, chlorophyll *a* (as a measure of phytoplankton primary productivity), dissolved oxygen, salinity and temperature are useful parameters for assessing eutrophication. Other indications of eutrophication involve measurements of changes in biota over time, such as areal coverage, distribution and condition of seagrass and macroalgal habitats, as well as species shifts in microorganism and macroalgal populations. Nutrient monitoring programs have been conducted both historically (1973-1981) and more recently (1988-1996) in the Great Bay Estuary by UNH researchers, and as part of the Seabrook Station Environmental Studies in Hampton Harbor by Normandeau Associates, Inc. Additionally, nutrient concentrations have been included in studies of nonpoint source pollution in the Great Bay Estuary (Jones and Langan 1993a; 1994a, b, c; 1995a, b, c; 1996a, b, c), and as part of a project assessing contamination of groundwater and surface waters by on-site sewage disposal (septic) systems in Seabrook and Hampton, NH (Jones et al., 1995, 1996). The monitoring and research studies are discussed here relevant to nitrogen, and to a lesser extent phosphorus, concentrations in New Hampshire estuaries.

2.4.1 NUTRIENT CONDITIONS IN NEW HAMPSHIRE'S ESTUARIES

The issue of nutrient overenrichment has been addressed in the Great Bay Estuary through monitoring programs dating back to the early 1970s as well as more recently in targeted studies of point and nonpoint nutrient inputs. Some of the data includes measures of organic nitrogen and phosphorus, however, the most temporally and spatially expansive data sets include inorganic forms of nitrogen (NH_4, NO_2^+ NO_3^-) and phosphorus (PO_4), forms which are most readily available for use by primary producers.

The Great Bay Monitoring Program supported by the GBNERR has included measurement of inorganic nitrogen and phosphorus concentrations at three sites in the Great Bay Estuary (Langan and Jones, 2000). Sites in the tidal portion of the Squamscott River and at Furber Strait (junction of Little Bay and Great Bay) have been sampled at high and low tide since 1988, while a site in the Lamprey River has been sampled since 1992. Though spatially somewhat limited, these data provide an excellent database from which short term changes in nutrient concentration can be detected. In addition, a substantial database generated between 1973-1981, which includes data from the Furber Strait/Adams Point site, allows for longer term trend analysis. The state shellfish program recently began monitoring shellfish growing waters for nutrients and other parameters, in addition to fecal indicator bacteria (Langan et al., 1999a).

Though concentrations differ between stations, the seasonal patterns are similar. Highest concentrations of inorganic nitrogen occur late fall through early spring, while the lowest concentrations occur in late spring through early fall. The seasonal pattern for PO_4 is somewhat similar, though following an initial drop during spring phytoplankton blooms, phosphate concentration often rebounds in summer. The timing of the spring phytoplankton bloom can vary considerably, depending on annual weather conditions, therefore the drop in N and P concentration can occur from late March to mid-May. At the Furber Strait site, maximum dissolved inorganic nitrogen (DIN=NH_4 + NO_3 + NO_2) can be as high as 20 µM in winter months, while minimum concentrations are generally < 1 µM at times in the spring and summer. Annual mean DIN at this site ranged from 7-11 µM from 1988 to 1996, with an eight-year mean of 8.8 µM. Interannual variation has been considerable and no long-term trend in concentration from 1988-1996 has been observed. Orthophosphate at Furber Strait has ranged seasonally from <0.10 µM to 1.5 µM with the annual mean ranging from 0.70 µM to 1.0 µM. The eight year mean is approximately 0.85 µM. Though at times the N:P ratio can range from as high 40:1 to as low as 1:1, the long term mean N:P ratio at this site is ≈ 10.6:1, indicating possible nitrogen limitation when compared to the Redfield ratio of 16:1. High tide concentrations of nitrogen at this site are slightly higher than at low tide, though this difference is inconsistent and statistically not significant. Orthophosphate concentrations are similar at high and low tides.

At the Squamscott River site (Chapman's Landing), nitrogen concentrations are much higher than at Furber Strait. DIN concentrations at this site can reach 40 µM during the winter and are generally <5 µM in spring and summer. The rapid drop in nutrient concentration in spring measured at Furber Strait is not as dramatic in the Squamscott River station, as spring turbidity, resulting from spring winds and freshwater runoff, often limits phytoplankton production. Therefore, nitrogen concentrations do not reach minimum concentrations until summer. The annual mean DIN from 1988 to 1996 at this site is ≈ 20 µM. DIN concentrations are generally higher in low tide samples, indicating an upstream riverine source of nitrogen in the Squamscott River. As was the case with the site at Furber Strait, there is considerable interannual variation in DIN concentration, though significant differences between years and trends in concentrations have not been evident in the eight year period. Orthophosphate concentrations have ranged from <0.3 µM to nearly 2 µM, with the overall mean of ≈ 1.25 µM. Though the N:P ratio can vary widely during the year, the overall eight-year N:P ratio is approximately 11:1, indicating some degree of nitrogen limitation like that at Furber Strait.

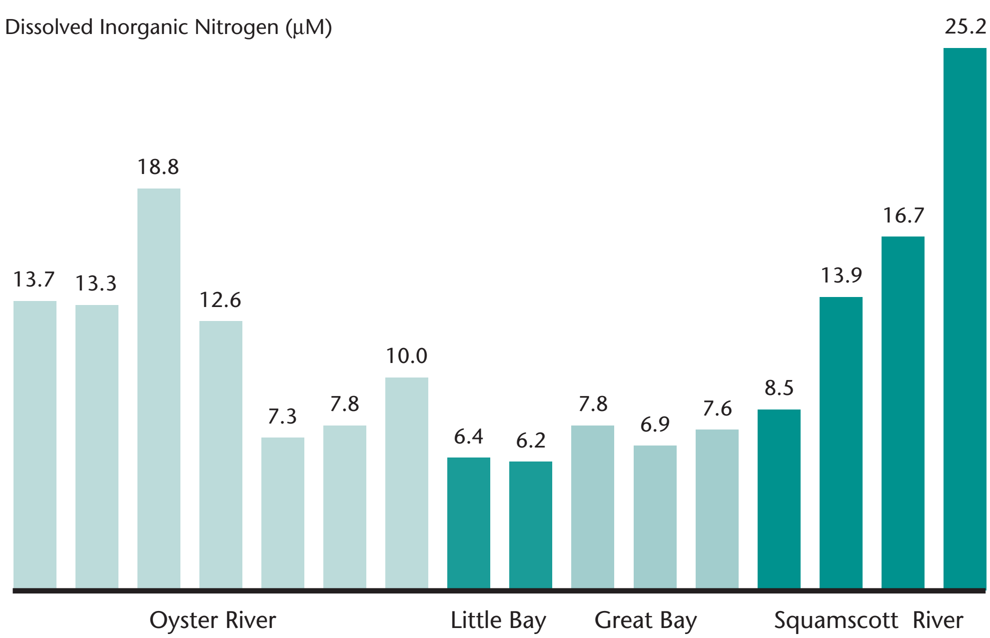
Nitrogen concentrations measured at the Lamprey River sample site are slightly higher than at Furber Strait, and lower than the Squamscott River. Concentrations of DIN can range from <1 µM to 30 µM, with annual means from 1992-1996 ranging from 10-14 µM. Orthophosphate is lower at this site than at the two other long term monitoring station, with a mean concentration of ≈ 0.6 µM. N and P concentrations at this site vary widely during the year, however, the mean ratio is ≈20:1.

Two separate field programs conducted concurrently from 1993 through 1995 (Jones et al. 1997) included measurements of nitrogen and phosphorus in samples taken on a transect beginning at the head of tide in the Oyster River, running south through Little Bay into Great Bay and terminating near the Newfields boat launch on the Squamscott River (Figure 2.29). Samples were taken monthly from a subset of stations with increased frequency at all stations during spring, summer and fall. Mean DIN concentration was highest at the station located at the Durham WWTP outfall in the Oyster River, and the influence of the treatment plant outfall was observed in the increased DIN concentration (18.8 µM) just downstream during low or falling tide. Otherwise, the highest concentration of DIN was measured at the most upstream site in the Squamscott River (25 µM), with decreasing concentrations (5-8 µM) through Great Bay into Little Bay. At the head of tide in the Oyster River, mean DIN was ≈ 13 µM, while at the mouth of the river, mean DIN was 10 µM. A short distance from the river mouth into Little Bay, mean DIN concentration (≈ 6 µM) was similar to Furber Strait and mid-Great Bay. Orthophosphate concentrations exhibited a similar pattern, with upstream stations as well as stations downstream of the Durham WWTF having the highest concentrations. Annual mean N:P ratios ranged from 7:1 to 11:1, indicating nitrogen limitation.

A three year project designed to assess the effect of storm events on concentrations of a suite of contaminants in the tributaries to Great Bay provided an excellent database for assessing spatial distribution of nutrient concentrations in the freshwater and tidal portions of the tributaries (Jones and Langan, 1994a, 1995a, 1996a). In addition to the inorganic forms of nitrogen and phosphorus, particulate nitrogen was measured in year two of the study, and dissolved organic nitrogen was measured in years two and three. Sampling was conducted at the same sites used in Figures 2.6 and 2.7 during dry periods (no precipitation for five days prior to sampling) and during

FIGURE 2.29

Dissolved inorganic nitrogen (DIN) concentrations at sites along a transect from the Oyster River through Little and Great Bays to Newfields on the Squamscott River.



the first low tide occurring within 24 hrs of a rainfall event of 0.5" or more. In year one, eight dry and eight storm events were sampled, while in years two and three, four storms were sampled on two consecutive days following storms. In addition to the tributaries, years one and two included stations in Hampton Harbor and the lower Piscataqua River. Though consistent effects of rainfall events on nutrient concentrations were not found, the dataset provides an excellent record of the spatial distribution of nutrient concentrations and a means of evaluating nutrient loading from point and nonpoint sources. The highest nutrient concentrations were consistently found in the freshwater and tidal portions of the Cocheco and Salmon Falls rivers. Relative to other sites, nutrient concentrations were also elevated in the freshwater portions of the Oyster River and in the tidal portion of the Squamscott River. Nutrient concentrations were consistently low in Hampton Harbor and the Piscataqua River. Relative to the forms of nitrogen, particulate nitrogen was generally a small fraction of the total, and exceeded 10% of the total nitrogen only during phytoplankton blooms at some sites. Dissolved organic nitrogen (DON) concentrations often exceeded DIN concentrations, however, DON represented a smaller fraction of the total at sites with the highest combined nitrogen concentrations.

Nonpoint source pollution assessments in the Oyster and Squamscott Rivers (Jones and Langan 1994a,c; 1995a,c; 1996a) included measurement of inorganic nutrients at sites along the tidal mainstem of the two rivers, sites in the freshwater portions of the rivers, small streams entering both portions of the rivers, and adjacent to suspected pollution sources such as developments and agricultural sites. In the Oyster River, the highest concentrations of dissolved nitrogen and phosphate were found in the vicinity of the Durham WWTF outfall and immediately above the tidewater dam in the Mill Pond. The greatest influence on overall nitrogen concentration, however, was from the treatment plant. A nitrogen and phosphorus plume was detectable at upstream stations all the way to the head of tide during flood tides, and as far downstream as Johnson Creek and sometimes Bunker Creek during ebb tides. The high nutrient concentration from the WWTF plume made it difficult to determine the relative strength of other tidal sources. Samples taken upstream of the Mill Pond, in both the main stem of the river and in smaller tributaries such as College Brook and Pettee Brook frequently had higher nitrogen concentrations than the water coming over the dam. A similar situation was found in Beards Creek which has a small impoundment before reaching the tidal portion of the river. The data indicates that impoundments can potentially remove nitrogen either via uptake by phytoplankton and macrophytic aquatic vegetation, or by biogeochemical processes such as denitrification or burial.

In the Squamscott River, a trend of decreasing nutrient concentration was identified from the head of tide in downtown Exeter to the mouth of the River in southwestern Great Bay (Jones and Langan, 1995c). Freshwater concentrations of nutrients were lower than tidal concentrations, indicating that the primary sources of nutrients were downstream of the tidal dam and may include the Exeter WWTF, runoff from the urban portion of Exeter, overflow from a CSO impoundment, dairy farms such as the Stuart Farm in Stratham and possibly the Rockingham Country Club golf course. Elevated nitrogen concentrations at the mouths of some marsh creeks whose drainage was undeveloped indicated that marshes may be exporting nitrogen.

Water column nutrient concentrations in the lower estuary were measured as part of the Ecological Risk Assessment Study for the Portsmouth Shipyard (Langan, 1994). This project included an initial set of replicate samples taken at 21 stations in the Piscataqua River, followed by monthly samples taken at low tide for a two year period at a subset of six stations. Nitrogen concentrations followed a seasonal pattern similar to the upper estuary, with the highest concentrations occurring in late fall through early spring, and the lowest concentrations (0-1 μM) measured from late spring through fall. Annual mean DIN for the six stations on the harbor area ranged from ≈ 7-10 μM. The highest concentrations of NH_4 and NO_3 were measured in Cutts Cove, which receives ebb tide waters from North Mill Pond, and at the Sarah Long Bridge, close to the Kittery, ME shore, just downstream from the Kittery WWTF. Orthophosphate concentrations were similar at all stations with the annual means ranging from 0.6 to 0.8 μM and individual measurements ranging from 0.2 to 1.2 μM.

The Portsmouth Shipyard Risk Assessment project also included three fixed station tidal stage studies, four crosssectional transects and high and low tide longitudinal transects conducted in July 1993. Data from transects and fixed station studies in the lower river and at the mouth of the Harbor indicated that nitrogen concentrations were very low, and generally on the order of 0-1 μM regardless of tidal stage. All lower estuary samples had low PO_4 concentrations as well, ranging from 0.3 to 0.6 μM. Nitrogen concentrations were generally higher for the Dover Point crosssectional transect, with NO_2 + NO_3 ranging from 1-5 μM, and NH_4 concentrations ranging from 1-4 μM. The highest concentrations were measured in the upper Piscataqua River during mid-ebb tide, indicating an upstream source of nitrogen. Longitudinal transects beginning at the mouth of Portsmouth Harbor to the railroad bridge on the Squamscott River were conducted at high and low tides on consecutive days. NO_2 +NO_3 concentrations on the high tide transect ranged from 0-1 μM from the harbor mouth to Dover Point and from 1-2 μM from Dover Point to the Squamscott River. For the low tide transect, NO_2 +NO_3 concentrations were similar to those measured at high tide in the lower estuary, and with the exception of samples taken in the upper Piscataqua River and at the mouth of the Squamscott River, were slightly lower (0-1.5 μM) through Little and Great Bay. Ammonium concentrations were more variable for both tidal longitudinal transects, ranging from 0-5 μM. The lowest concentrations were measured in the lower Piscataqua River and upper Great Bay at both tides, while the highest concentrations were measured at low tide in the upper Piscataqua and Squamscott rivers. The longitudinal transect data indicates possible sources of nitrogen from these two general (upstream) sources. Orthophosphate concentrations, though low throughout, increased from the harbor mouth to the upper estuary at both tides, with concentrations ranging from 0.3 to 0.8 μM.

A study of the sanitary quality of the shellfish growing waters in Little Harbor (Jones and Langan 1995c) included measurement of nutrient concentrations at sites in the vicinity of the Wentworth by the Sea golf course. Samples were taken in the spring following fertilizer application and during a period of wet weather. Mean DIN concentrations at three sites ranged from 6.16 μM to 10.2 μM while mean PO_4 concentrations ranged from 0.32 to 0.49 μM.

Based on the studies reviewed for this document, some general statements can be made regarding temporal and spatial patterns of nitrogen and phosphorus concentrations in the Great Bay Estuary. Throughout the estuary, the highest nutrient concentrations occur in late fall through early spring and the lowest concentrations occur in late spring through early fall. This pattern is more well defined for NO_2 + NO_3 than for NH_4 and PO_4. Spatially, the highest nitrogen concentrations generally occur near the heads of tide, due either to freshwater influences (Cocheco, Salmon Falls, Oyster Rivers) or to the location of municipal WWTF outfalls near the heads of tide (Oyster River, Exeter/Squamscott River, Salmon Falls River). Spatially, phosphate concentrations are low in most of the freshwater portions of the tributaries, highest in the upstream portions of the tidal rivers, and lower through Great Bay, Little Bay and down to the harbor mouth. There is an inverse relationship of salinity with nitrogen concentration, with the lowest concentrations occurring in the lower Piscataqua and Little Bay. By comparison with nutrient concentrations in other estuaries in the Northeast U.S., the Great Bay Estuary probably falls somewhere in the middle of the field.

By comparison to the Great Bay Estuary, very little data on nutrient conditions exists for the Hampton/Seabrook Estuary. A long term dataset has been established by Normandeau Associates (NAI, 1996), however, only one station outside the Harbor has been monitored and the data do not accurately represent conditions in the estuary. As part of a two year study of the potential for groundwater and surface water contamination from septic systems (Jones et al., 1995; 1996), nutrients were measured in groundwater and surface water at sites in Seabrook and Hampton. At eleven sites in Seabrook, groundwater wells were sampled in and around the effluent disposal areas (EDA) of residential homes. Surface waters down gradient of the EDAs, which were either fresh or brackish streams, marsh creeks or the Harbor itself, were also sampled. DIN concentration in the wells ranged from 0.15 to 36 mg/L, while the annual mean DIN concentration in surface waters ranged from 0.06 mg/L in the mouth of the Harbor to 2 mg/L in some of the small freshwater creeks. There was a decreasing nitrogen concentration with increasing salinity for the surface water samples. Based on the nitrogen concentrations and the direction of flow determined in the hydrological studies, it appears that nitrogen is transported from EDA to surface water, however the resulting low nitrogen concentrations in the harbor and the absence of any signs of potential eutrophication (low dissolved oxygen, algal mats, extreme phytoplankton blooms, etc.) indicate that there is little observable impact to the estuary. Though phosphate was detected in high concentrations in and around the EDAs, it did not appear to be as readily transported in the groundwater to surface waters. PO_4 concentration ranged from 0.01 to 8.9 mg/L in the EDA and from 0.01 mg/L to 0.06 mg/L in surface waters. A follow-up study in 1996-97 showed nutrient concentrations in the same surface waters were not significantly different from previous years, even though septic systems were being disconnected throughout Seabrook (Jones, 1997).

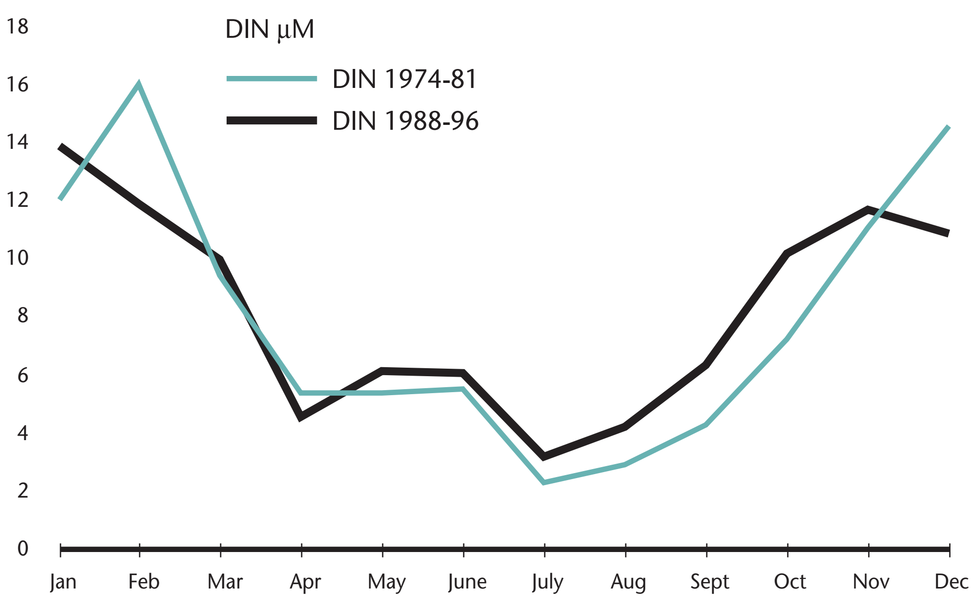
2.4.2 TRENDS IN NUTRIENT CONCENTRATIONS

Assessing long term trends in nutrient concentrations requires consistent sampling and analytic protocol over an extended period of time. Though some of the studies described above were conducted for two or three consecutive years, normal variation in water column concentrations makes it difficult to detect trends. Nutrient data generated for the Great Bay NERR Monitoring program, which has included sampling and analysis for eight years at two of the three stations indicates that there is considerable interannual variation in nutrient concentrations. However, statistical analysis of the eight years of data (ANOVA) does not indicate any significant differences in either nitrogen of phosphate concentrations between years nor are any trends of increasing or decreasing concentrations evident. The data collected as part of the Great Bay Field Program (Loder and Gilbert 1977; 1980; Loder et al., 1983; Daley et al., 1979; Norall, et al., 1982) included low tide sampling and analysis at stations that included a site at Furber Strait, identical to the 1988-1996 site sampled in the GBNERR monitoring program. Analytical methods for the earlier and more recent datasets were not identical, however, they were sufficiently similar to enable comparisons of nutrient concentrations. When all compatible (depth sampled) data for the earlier and more recent datasets were compared, mean NH_4 concentration was slightly higher in 88-96 dataset (3.51 µM) than in the 1973-1981 dataset (2.57 µM). Conversely, mean NO_2 $+NO_3$ concentration was slightly lower from 1988-1996 (5.25 µM) than 1973-1981 (5.60 µM). Mean dissolved inorganic nitrogen (NH_4 + NO_2 $+NO_3$) at the Furber Strait site is therefore slightly higher from 1988-96 (8.76 µM) than from 1973-1981 (8.17 µM). The datasets were compared statistically using both parametric (t-test) and nonparametric methods and no significant difference in DIN concentration was found. Seasonal patterns were also analyzed. There was considerable variation between years for samples taken during a particular month, therefore monthly means for the earlier and recent datasets were used for the purpose of comparison. The seasonal patterns for NH_4, NO_2 $+NO_3$ and DIN for the two datasets were remarkably similar to the data for DIN presented in Figure 2.30. As was the case when all data were compared, monthly mean NH_4 concentrations were slightly higher in the more recent dataset, and NO_2 $+NO_3$ were slightly lower.

Two additional studies conducted in 1976-1977 (Daley and Mathieson, 1979; Loder et al., 1979) allow an evaluation of changes in riverine nitrogen concentrations over a nearly 20-year period. Hourly water samples were collected throughout full tidal cycles in July and August in 1976 and 1977 (Daley and Mathieson, 1979) immediately seaward of the tidal dams and at sites downstream of the tidal dams and analyzed for NO_2 $+NO_3$. The mean concentrations were compared to July and August means for equivalent sample sites collected for various studies from 1993-1996. These data are presented in Figure 2.31 and 2.32. Increased concentrations over the nearly 20 year period are observed in the freshwater sites in the Cocheco and Salmon Falls rivers (Figure 2.31) while nitrite-nitrate concentrations are lower in the freshwater and estuarine portions of the Oyster and Bellamy Rivers (Figure 3.32). Similar concentrations for the two periods were observed in the Lamprey and Squamscott rivers.

Monthly data were collected and analyzed for nitrate-nitrite at the terminal freshwater areas of the Great Bay tributaries from February 1976 through June 1978 as part of study on nutrient flux processes in the estuarine system (Loder et al., 1979). Sample means were calculated and compared to data collected for several studies at identical sites from 1993-1996 (Jones and Langan, 1996a; Langan and Jones, 1996; Jones et al., 1997). The results are similar to the July-August data comparisons. Nitrate-nitrite concentrations at all sites with the exception of the freshwater areas of the Cocheco and Salmon Falls rivers are either similar to or lower in the more recent dataset, indicating improvements or no change in all tributaries except the Salmon Falls and Cocheco rivers, where concentrations have increased. Statistical

Monthly mean dissolved inorganic nitrogen at Adams Point in Great Bay for the years 1973-81 and 1988-96. **FIGURE 2.30**



analysis (t-tests as well as nonparmetric tests) indicate significantly higher concentrations of nitrate-nitrite in the freshwater portions of the Cocheco and Salmon Falls rivers, significantly lower concentrations in the freshwater and estuarine portions of the Oyster and Bellamy rivers, and no significant differences for the Lamprey and Squamscott rivers between data from the mid-1970s and the mid-1990s.

Based on the data reviewed for this report, it is possible to make some general statements regarding trends in nutrient concentrations in the Great Bay Estuary. Despite a dramatic increase in population from 1970 to 1990 (and a slower increase since 1990) throughout the Great Bay watershed, and therefore an expected increase in nitrogen loading,

FIGURE 2.31 *Nitrate/nitrite concentration trends in freshwater portions of tributaries to the Great Bay Estuary.*

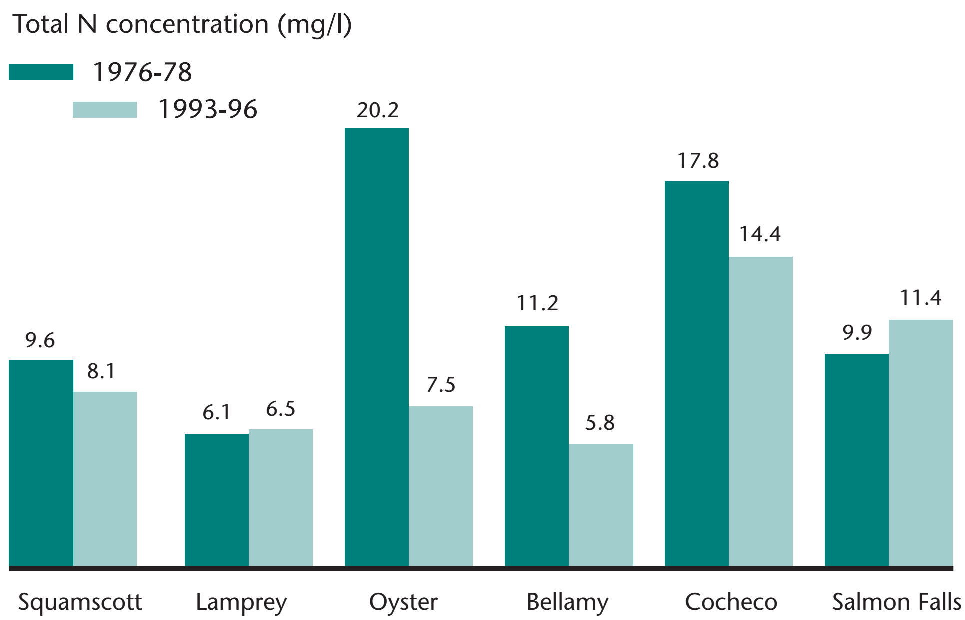
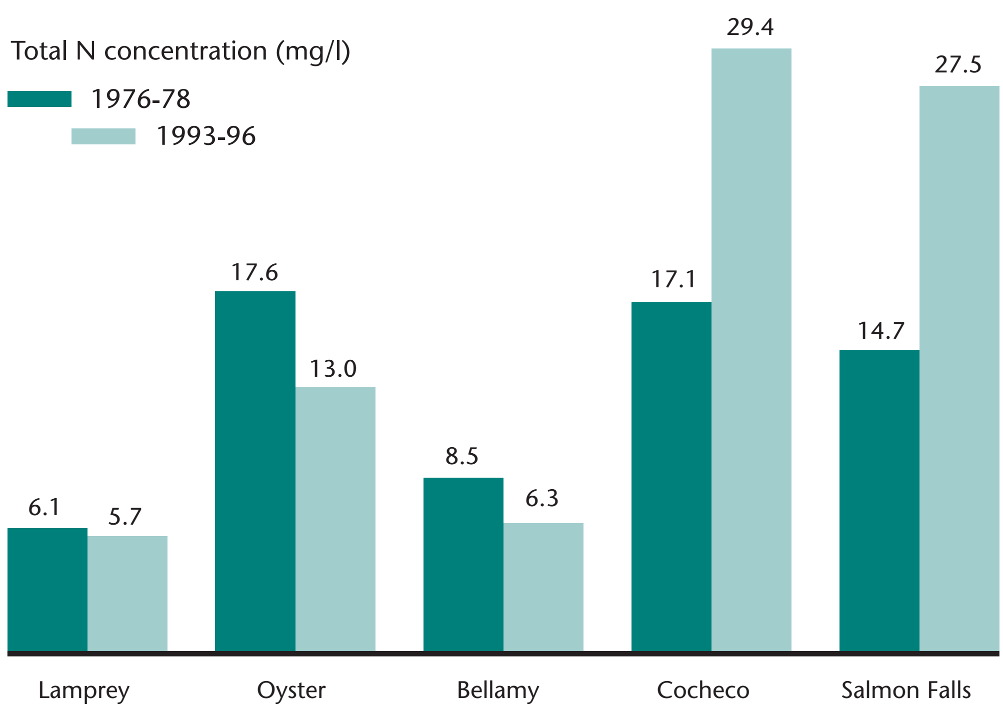


FIGURE 2.32 *Nitrate/nitrite concentration trends in saltwater portions of tributaries to the Great Bay Estuary.*

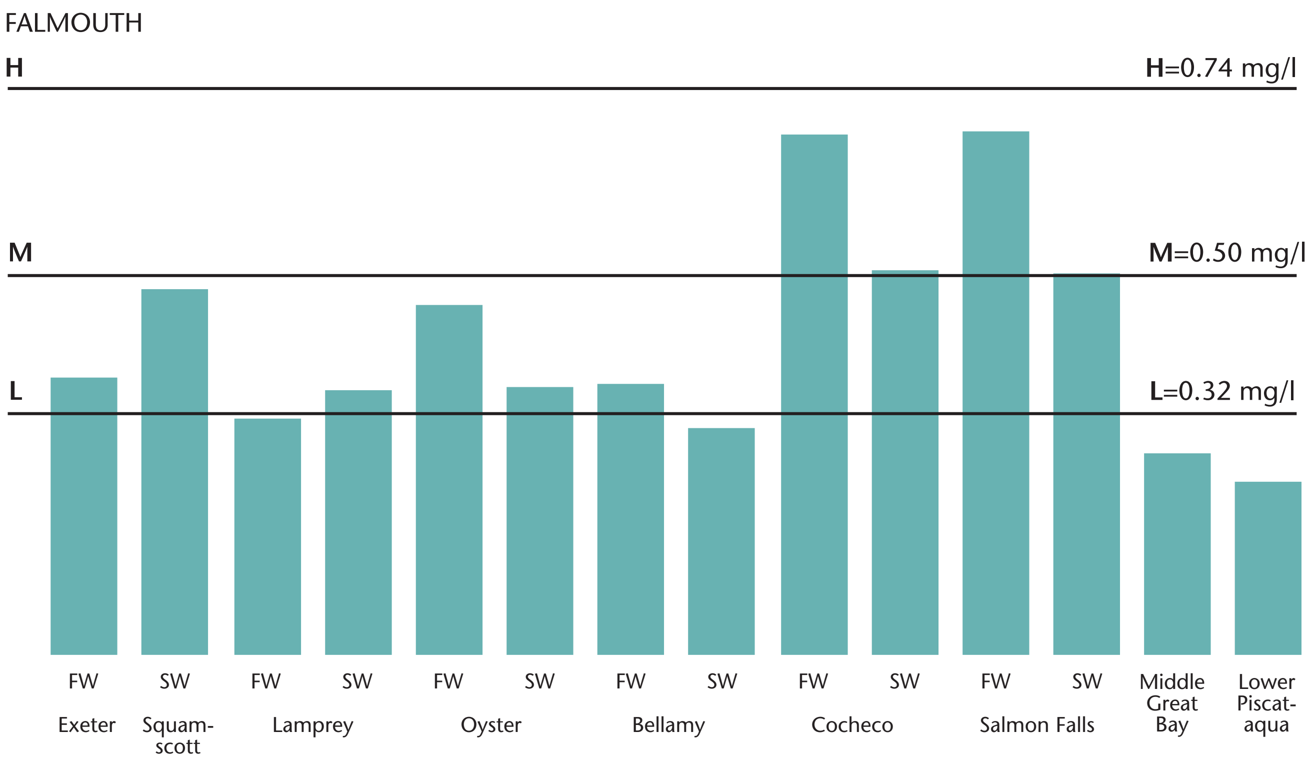


recent data indicate that current nutrient concentrations (annual means, seasonal patterns, minimum and maximum concentrations) in most areas of the estuary, including the tidal tributaries are similar to or lower than that which was observed in the 1970s. The exceptions are the Cocheco and Salmon Falls rivers, and in particular the freshwater portions of those rivers, where concentrations have increased in recent years. One possible explanation is that the expected increased loading from increased population has been offset by improvements in municipal wastewater treatment in most areas.

2.4.3. RELATIONSHIP TO WATER QUALITY STANDARDS

Though water quality criteria for estuarine waters have been established for some parameters such as metals, fecal indicator bacteria and dissolved oxygen, examples of concentration limits for nitrogen are rare. The Town of Falmouth, Massachusetts (1994) adopted a three tiered nitrogen concentration approach intended to limit future nitrogen inputs. Total nitrogen concentrations of 0.32, 0.5 and 0.75 mg/L total N were established as critical concentrations for water bodies of varying usage and classifications. Though the Great Bay Estuary has different characteristics than water bodies in the Town of Falmouth, it is useful to compare nitrogen concentrations in Great Bay to the standards established for Falmouth. Total nitrogen data for Great Bay locations were obtained from several studies described above, including the three year study of the tributaries (Jones and Langan (1994a, 1995a and 1996a) and data from a nonpoint source assessment extending from Oyster River through Squamscott River (Jones et al., 1997). Results are presented in Figure 2.33. None of the mean concentrations of total N, including the freshwater portions of the Cocheco and Salmon Falls rivers, exceed the 0.75 mg/L upper limit set for Falmouth. Sites exceeding the Falmouth medium concentration criteria (0.5 mg/L) include both the freshwater and tidal portions of

Comparison of total nitrogen concentrations for Great Bay Estuary and its freshwater and estuarine tributaries with Falmouth, MA water quality benchmarks. **FIGURE 2.33**



the Salmon Falls and Cocheco rivers. Sites exceeding the Falmouth low limit (0.32 mg/L) include the freshwater and tidal sites in the Exeter/ Squamscott River, the tidal sites in the Lamprey and Oyster rivers, and the freshwater site in the Bellamy River. Sites in the freshwater portion of the Lamprey River (0.30 mg/L), Little Bay/Bellamy River (0.29 mg/L) mid-Great Bay (0.27 mg/L) and the Piscataqua River (0.23 mg/L) are all lower than the Falmouth lower limit of 0.32 mg/L. The Great Bay Estuary could generally be characterized as having higher turbidity, greater flushing and greater depth than the water bodies surrounding Falmouth, therefore it is likely that it is less sensitive to higher nitrogen concentrations (Nixon and Pilson 1983).

2.4.4 POLLUTION SOURCES AND NITROGEN LOADING ESTIMATES

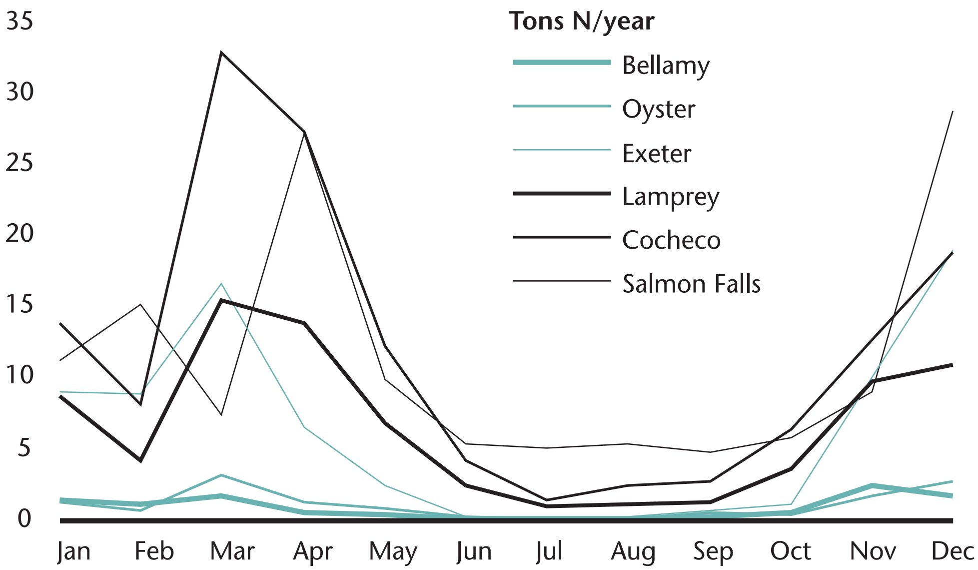
In general, sources of nutrients to estuaries include natural sources such as watershed sediments, organic debris (leaves and other vegetation) and groundwater, as well as point and nonpoint sources of anthropogenic origin. Anthropogenic point sources include industrial and municipal wastewater while nonpoint sources include urban and agricultural runoff, stormwater conduits, on-site wastewater treatment (septic) systems, lawn fertilizers and atmospheric deposition of nitrogenous compounds that result from burning of fossil fuels.

Loading estimates to water bodies are frequently based on modeling exercises. Values for nitrogen contribution, either measured from previous studies or estimated from literature values, can be assigned to all types of land use and cover (urban, forested, wetland, active agriculture, lawns, impervious surfaces), population and method of waste disposal in a watershed. Coupled with meteorological (rainfall) and other physical data (soil type, river discharge) the land use and land cover data can be used to estimate annual loading of nutrients. The NOAA Status and Trends Branch (NOAA, 1989), estimated annual loading to the Great Bay Estuary of 636 tons of nitrogen and 204 tons of phosphorus. Of these totals, it was estimated that point sources are responsible for 242 tons of nitrogen and 161 tons of phosphorus, while nonpoint sources are responsible for 394 tons of nitrogen and 43 tons of phosphorus. The method used to make these estimates is unclear, but it is assumed that it was some type of modeling study based on satellite derived (GIS at 1:24,000) land use/land cover data and predetermined values for nitrogen contribution. Another NOAA publication from the Strategic Assessment Branch (NOAA, 1994) estimated the total nitrogen input from point sources to be 317 tons per year. This estimate was based on effluent volume monitoring and typical wastewater concentrations of nitrogen.

Sources in Great Bay include municipal wastewater treatment plants, septic systems, urban and suburban (lawn fertilizer) runoff, and atmospheric deposition. Though agriculture is often cited as a major source of nutrients to estuaries, this is probably not the case in Great Bay. Though some farms may input nutrients at specific locations (i.e., Aikman Dairy Farm on the Salmon Falls River and Stuart Farm on the Squamscott River) there is very little active agriculture in the watershed, and therefore little possibility for system-wide loading of nutrients from agricultural sources. The models that use current GIS data to estimate nutrient loading may tend to overestimate the contribution of agriculture, since some of the land identified as active agriculture has not been farmed for many years. Additionally, some of the larger farms adjacent to the estuary (those mentioned above) have recently adopted, with the assistance of the NH Coastal Program and the Natural Resource Conservation Service (NRCS), best management practices to reduce contamination from animal wastes and fertilizer application.

The numerous studies on nutrient concentrations described in the earlier section of this report, in addition to studies on streamflow and river discharge (Pappas, 1996), atmospheric deposition (Mosher, 1995), and on effluent quality from local sewage treatment plants (Mitnik, 1994) have made it possible to esti-

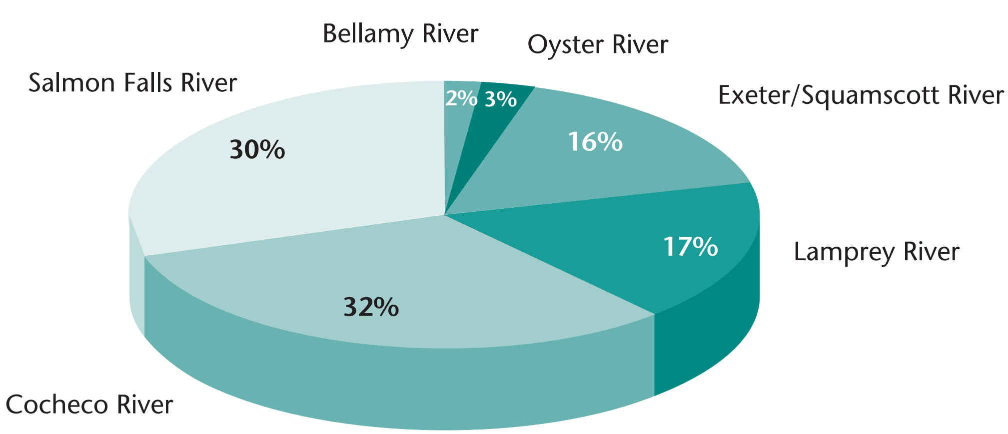
Annual loading of nitrogen from fluvial (riverine) sources to the Great Bay Estuary. **FIGURE 2.34**



mate loading to the Great Bay Estuary from actual measured data. There is also some data available on urban stormwater (Jones, 1998b; Jones and Langan, 1996a), however most of the urban development in the NH Seacoast is located at the heads of tide, and most stormwater is diverted to the freshwater portions of the tributaries and would therefore be included in the fluvial (riverine) loading estimates. For the purposes of this report, this exercise was limited to nitrogen, since it has been identified as the limiting nutrient in most estuaries, including Great Bay.

Fluvial (riverine) loading, which includes both natural and anthropogenic sources, was calculated by using mean monthly concentrations of total nitrogen (DON + DIN + PN) measured over a three year period in the tributaries to Great Bay (Jones and Langan 1994a, 1995a, 1996a) and river discharge measured and calculated by Pappas (1996). These data are presented in Figure 2.34. Nitrogen loading estimated for tributaries to the tidal portions of the Oyster River (Jones and Langan 1993a, 1994c) and Squamscott River (Jones and Langan 1995c) were small (on the order of < 1 ton annually from all tributaries) by comparison to the main stem of each river and to WWTFs, and were therefore not used in the calculations. Throughout the year, the months with the greatest loading are understandably the months of great-

Nitrogen loading to the Great Bay Estuary from fluvial (riverine) sources. **FIGURE 2.35**

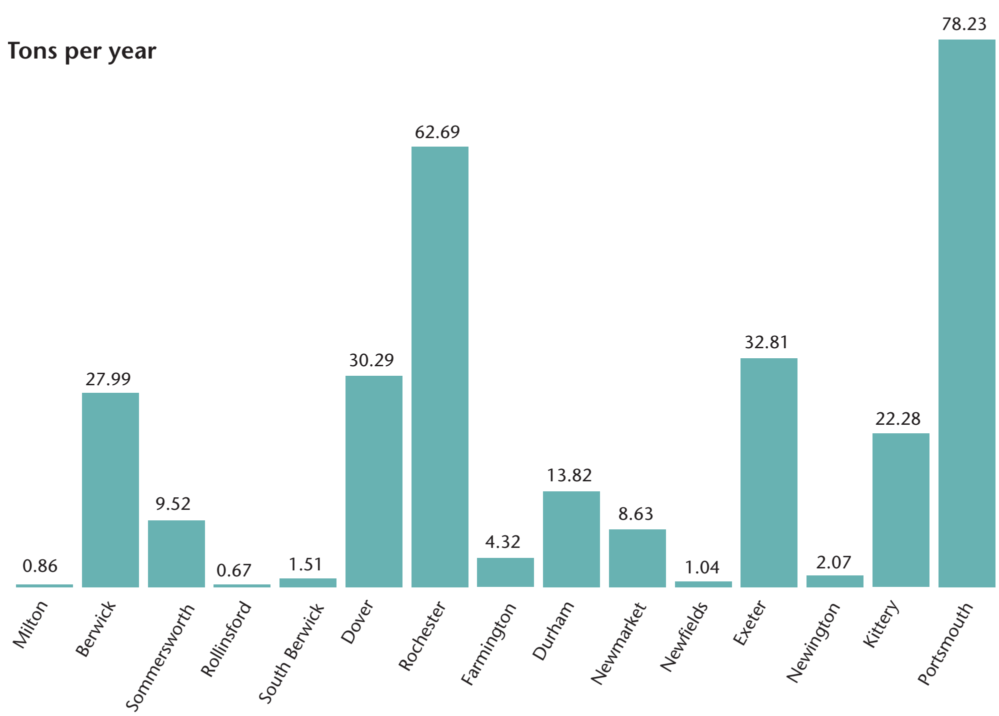


est river discharge. Peaks in loading occur in March and April and in November and December (Figure 2.34). Riverine nitrogen contribution to the Great Bay Estuary is greatest from the Cocheco and Salmon Falls rivers, followed by the Exeter and Lamprey rivers, with the smallest amount from the Oyster and Bellamy rivers (Figure 2.35). Nitrogen loading in the summer, or during dryer periods of the year, is greatest in the Salmon Falls River, followed by the Cocheco and Lamprey rivers. On an annual basis each river contributes the following in tons of N and % of total: Cocheco 143 (32%); Salmon Falls 134 (30%); Lamprey 78 (17%); Exeter 74 (30%); Oyster 12 (3%) and Bellamy 9 (2%) for a total of 450 tons of nitrogen per year.

Point source contribution was calculated using total nitrogen concentrations measured in wastewater effluent from the Milton, Berwick, South Berwick, Somersworth, Rollinsford and Dover WWTFs (Mitnik 1994) and the Durham WWTF (Jones and Langan 1994c) and average effluent volume reported by the treatment plants. For those plants where nitrogen concentration was not measured, a mean nitrogen concentration calculated from the treatment plants with measured data were applied. Point source loading from municipal WWTFs is presented in Figure 2.36. The largest nitrogen input, in descending order, is from the Portsmouth, Rochester, Dover, Exeter Berwick and Kittery WWTFs. Even though the volume from the Berwick plant is relatively small, the nitrogen contribution is high due to high nitrogen (especially ammonium) concentration in the effluent. From these data, it is estimated that the total point source (WWTF) contribution of nitrogen to the Great Bay Estuary is 296 ton of nitrogen per year. This figure is greater than the 1990 NOAA estimate of 242 tons and slightly less than the 1994 NOAA estimate of 317 tons, although it does not include loading from six industrial NPDES dischargers to the Estuary (Table 2.1).

In order to calculate point and non-point nitrogen loading, nitrogen contribution from treatment plants upstream of the tidal dams (Farmington and Rochester on the Cocheco River; Milton, Berwick, Somersworth and Rollinsford on the Salmon Falls River) was subtracted from the annual fluvial loads calculated for the rivers. This results in a total of

FIGURE 2.36 *Nitrogen input to the Great Bay Estuary from municipal wastewater treatment plants.*

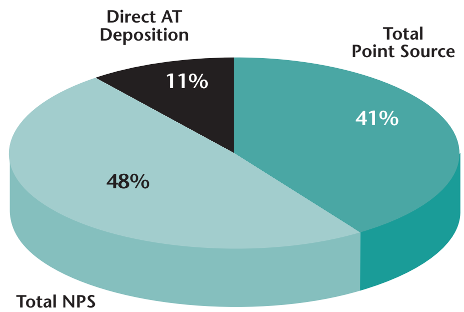


296 tons/year from municipal point sources, and 345 tons per year from fluvial sources (nonpoint sources).

Atmospheric deposition was calculated by Mosher (1996) for the Great Bay watershed. Since nitrogen loading from land deposition would be included in the fluvial source estimates, only direct deposition (to the water surface) was considered. The estimate for direct deposition was 77 tons/yr, which in addition to the point and nonpoint loading, totals 718 tons per year of nitrogen. The percentage contribution from the three sources is 48% from nonpoint sources, 41% from point sources and 11% from direct atmospheric deposition (Figure 2.37). The 718 tons per year is slightly greater than the 640 tons per year estimated by the NOAA Strategic Assessment Branch in 1990. In a smaller study conducted as part of a nonpoint sources assessment of the Oyster River in 1994, remarkably similar results with regard to the ratio of point and nonpoint contributions were obtained. Data generated by that study (Jones and Langan 1994c) estimated that 42% of the nitrogen loading to the Oyster River was from the Durham WWTF which contributed approximately 11 tons of total N per year.

It should be noted here that some liberties were taken in assignment of nitrogen inputs as either point or nonpoint. It is unlikely that the entire nutrient load from sewage treatment plants located well upstream of the estuary (Farmington, Rochester, Milton, etc) is delivered to the estuary. Therefore, attributing all of the nitrogen from these plants to point sources may result in an overestimate of point source contribution, and an underestimate of nonpoint source contribution. The total would not differ, however, since nonpoint was determined by subtracting the nitrogen contribution of upstream WWTFs from the total fluvial load. On another note, including the entire annual nitrogen contribution of the Portsmouth WWTF to estuarine loading may overestimate actual nitrogen loading to the estuary. The subsurface diffuser on the discharge pipe ensures rapid dilution, and the location of the outfall (near the mouth of the harbor), plus the characteristics and residence time of the receiving waters makes it unlikely that all or most of the nitrogen is transported upstream to the estuary, and that possibly up to 50% of the nitrogen is carried out of the estuary into the Gulf of Maine.

Sources of nitrogen loading to the Great Bay Estuary. **FIGURE 2.37**



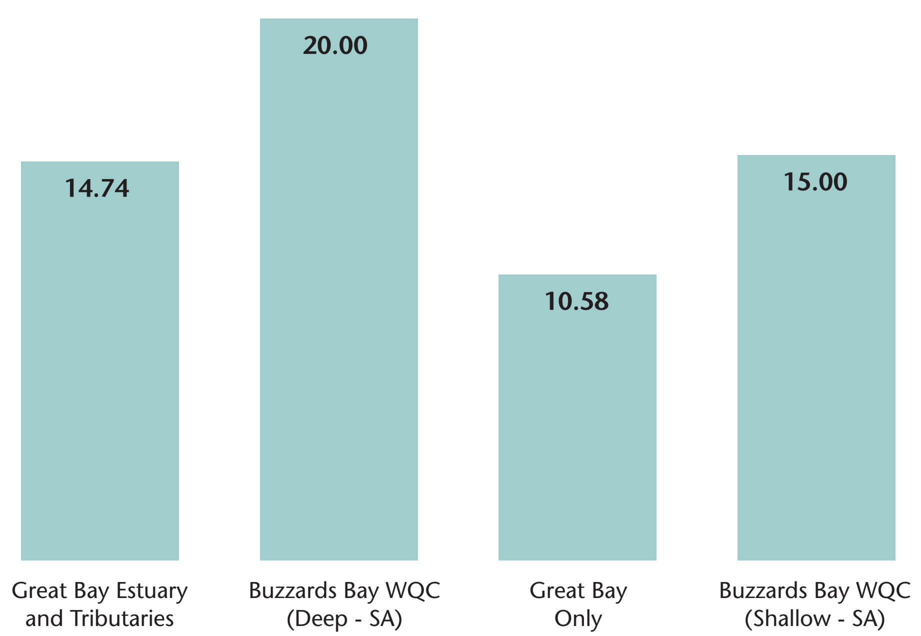
Although nonpoint (riverine) and atmospheric sources exceed point source inputs of nitrogen, these sources include natural as well as anthropogenic sources. Point sources (WWTFs) on the other hand, are almost entirely of anthropogenic origin. Therefore, loading from these sources becomes much more important when planning for future development and if it becomes necessary to consider nutrient reduction strategies.

As was the case with nutrient concentrations, nitrogen loading limits have not been established for the Great Bay Estuary. The State of Maine DEP (Mitnik and Valleau, 1996; Mitnik, 1994) has conducted a WASP modeling and Total Maximum Daily Limit study (TDML) on the Salmon Falls River, and found that there are nitrogen and phosphorus impacts (excessive phytoplankton and depressed oxygen) in the freshwater impoundments, and phytoplankton impacts (depressed oxygen) to a small portion of the tidal section of the river during dry periods in summer. This study will be discussed in the section detailing impacts of eutrophication.

The Buzzards Bay NEP established loading limits (expressed in g/m2 of water surface area/year) for anthropogenic nitrogen to the estuary. Similar to the Falmouth, MA concentration limits, a tiered approach to nitrogen loading was established depending on the depth and flushing characteristics of sections or subunits (subwatersheds of Buzzards Bay). Loading per unit area to the Great Bay Estuary was determined by using the estimates previously described (718 tons), and dividing by the surface area of the estuary (10,900 acres). The results were compared to the loading limit established for deep, SA (class A waters) in Buzzards Bay with a flushing time of >5 days. This would represent an average estimate for the Great Bay Estuary, since the depth range is very broad, and flushing time can range from hours to weeks, depending on the exact location in the estuary. Loading to Great Bay (Lower Little Bay and all of Great Bay) was also calculated, using the area (approximately 5,000 acres) and loading from the Exeter, Lamprey, and Oyster rivers (fluvial) and WWTFs in Exeter, Newfields, Newmarket and Durham. Direct deposition of nitrogen from atmospheric sources in proportion to the surface area was also considered. The Buzzards Bay limit for shallow class A waters with a flushing time > 5 days was used for comparison. Results of these calculations and comparison to loading limits established for Buzzards Bay are presented in Figure 2.38. Loading to the entire Great Bay Estuary was calculated to be 14.5 g/m2/year and loading to Lower Little Bay and Great Bay was calculated to be 10.4 g/m2/year. Both these figures are below the 20 g/m2/year for deep water and 15 g/m2/year for shallow water established for Buzzards Bay.

It must be stated, however, that these estimates are a first attempt to assess the nitrogen loading to the Great Bay Estuary from actual water quality data. Since loading was based on mean nitrogen concentrations, which can be highly variable in riverine waters as well as in wastewater, there is a degree of uncertainty for those areas where sample size was small or where the effluent concentration was estimated. The contribution of nitrogen from groundwater sources directly to the estuary is unknown. Though soils in the Great Bay Estuary differ from those estuaries that have significant input of nitrogen from groundwater (Buttermilk Bay and Waquoit Bay, MA), it may be possible that additional nitrogen loading occurs through direct groundwater input to the estuary. Since groundwater loading is not considered, this could result in an underestimate of the total

FIGURE 2.38 *Comparison of nitrogen loading in the Great Bay Estuary with water quality criteria standards established for Buzzards Bay, MA.*



loading. There is also a degree of uncertainty in the validity of Great Bay to Buzzards Bay comparisons due to differences in hydrographic condition, watershed geology and topography. Mean tidal height at the mouth of the Great Bay Estuary is approximately 2.7 meters, considerably greater than in Buzzards Bay (1.7 meters), and there is also greater mean water depth in some sections of the Great Bay Estuary. Though these differences would suggest that the Great Bay Estuary can handle a greater amount of nitrogen loading than Buzzards Bay, the uncertainties mentioned, in addition to the absence of a nitrogen budget for the Great Bay Estuary that includes accurate estimates of rates of nitrogen processes (uptake, burial, remineralization, denitrification), would make a definitive statement of that nature premature. Also, the limitations for Buzzards Bay were for anthropogenic nitrogen, whereas all sources of nitrogen were considered for the Great Bay analyses.

Nutrient loading has not been estimated for the Hampton/Seabrook Estuary. Sources of nutrients include groundwater contaminated by septic systems, the Hampton WWTF located on Tide Mill Creek, some small amount of active agriculture, and urban and suburban stormwater runoff. Hampton Harbor is quite unique in that it receives an 88% exchange of water on each tide (twice daily). Therefore, the residence time of the water in the estuary is on the order of hours, even for the upstream areas. This residence time is probably too short to support intense phytoplankton blooms, and indeed there is no evidence of these occurring (Jones, 1997). The nitrogen concentrations measured in the estuary and outside the harbor mouth (NAI, 1996) indicate that despite the probability that the estuary receives nitrogen input from point (WWTF) and nonpoint sources (septics, stormwater, etc.), there appears to be sufficient dilution to reduce concentrations of nitrogen to low levels. The absence of other indicators of nutrient overenrichment such as poor water clarity, low dissolved oxygen, dense macroalgal mats and proliferation of opportunistic algal species supports the finding that excess nutrient input is not a problem in Hampton Harbor. Additionally, the town of Seabrook has recently finished the process of linking all the residences to a centralized municipal sewage system. The outfall for the WWTF is located in the Atlantic Ocean, therefore the possibility of any impact from contaminated groundwater (from septic systems) will be permanently removed.

2.4.5. DOCUMENTED IMPACTS ON WATER CHEMISTRY AND NATURAL RESOURCES

The biological effects of nutrient enrichment can range from subtle to extreme. Species shifts in phytoplankton communities can result in unfavorable conditions for estuarine biota, particularly for filter feeders such as bivalve molluscs. Massive blooms of phytoplankton can reduce water clarity, shade submerged aquatic vegetation (SAV), and reduce water column oxygen concentration in the dark via respiration. Blooms of nuisance macroalgae can replace more desirable forms of vegetation and create hypoxic or anoxic conditions that can impact fish and invertebrates. Conditions resulting from nutrient enrichment can affect recreational activities such as fishing, boating and swimming as eutrophic systems can be most unappealing for these activities.

2.4.5.1 Dissolved Oxygen

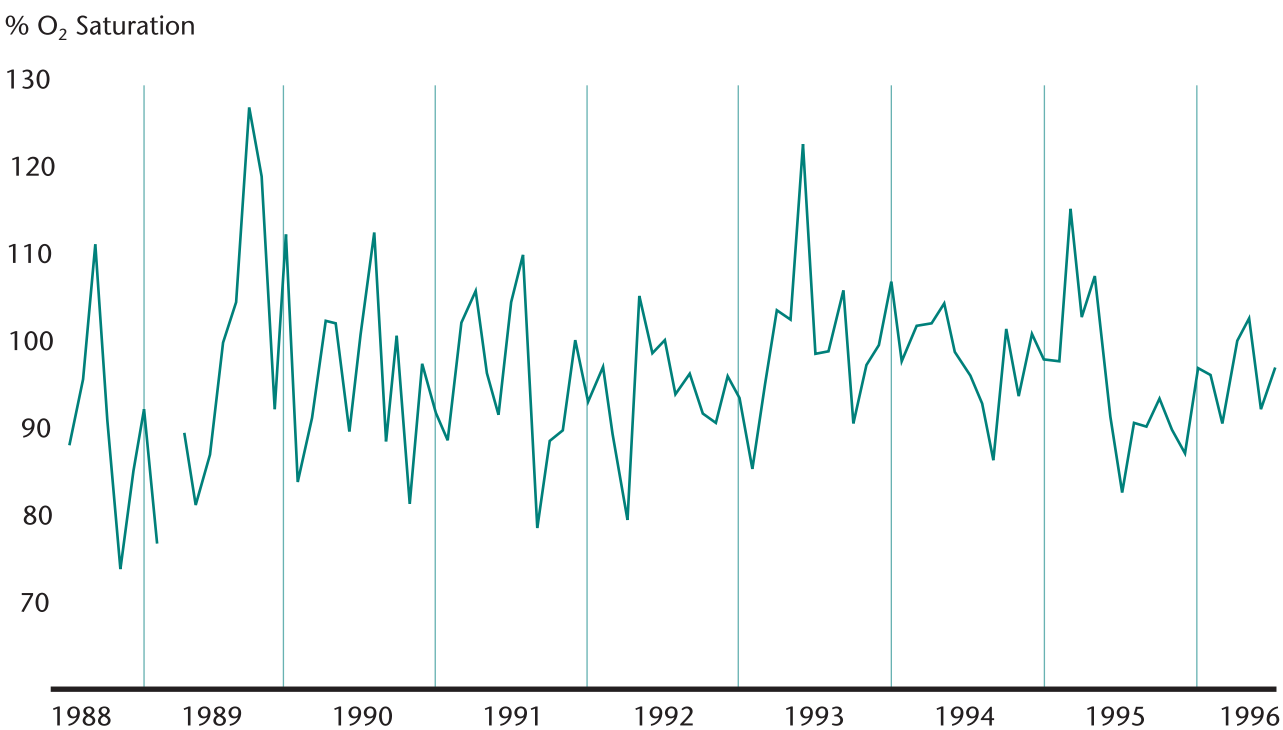
One of the principal concerns associated with nutrient overenrichment and eutrophication is reduction in dissolved oxygen (D.O.) due to elevated aerobic metabolism. Low D.O. (hypoxia) or the total absence of D.O. (anoxia) can severely impact aerobic marine and estuarine organisms and threaten the vitality of aquatic ecosystems. Dissolved oxygen is an important indicator and one of a suite of ecological endpoints for eutrophication.

Dissolved oxygen has been measured in association with many monitoring and research programs. In the Great Bay Estuary, dissolved oxygen can vary at all times of the year depending on temperature of the water. Colder, fresher water, has a great capacity for dissolved oxygen. Therefore, in winter, dissolved oxygen will be higher in the upper reaches of the estuary than in the more oceanic lower portions of the estuary. As the waters warm and salinity increases in summer in the upper estuary, dissolved oxygen will be lower than in the cooler lower estuary. Thus, the annual variation is expected to be greater in the upper tidal reaches of the estuary. Dissolved oxygen concentration is also affected by the depth of the water, the amount of mixing, residence time of the water, tidal stage and at certain times of the year, the time of day.

Though the absolute value of dissolved oxygen (measured in mg/l) is important, the degree or percent of oxygen saturation is a more accurate measure of the potential for biological effects. In general adverse biological effects are not evident unless dissolved oxygen drops below 5 mg/L for an extended period of time. The State of New Hampshire has established 75% saturation as the water quality standard for D.O. for not less than 16 hours per day and not less than 6 mg/l at any time except as naturally occurs. It is suspected that some shallow upper estuarine systems may drop below 75% saturation in the absence of eutrophication related impacts (Kelly, 1995).

Even though sites in mid-Great Bay can have dissolved oxygen ranging from 6 to 15 mg/liter throughout the year, percent oxygen saturation is usually between 90-110% (Figure 2.39) (Langan and Jones 1996). Lower estuary measurements vary similarly and are almost always near 100% saturation (Langan, 1994). Water column measurements indicate that there is little stratification and that dissolved oxygen is similar in value and percent saturation throughout the water column. In the tributaries to Great Bay, dissolved oxygen can vary from 5 mg/l during early morning low tides in summer to 16 mg/l in winter. Percent saturation in the Squamscott River, for example, can range during the year from

FIGURE 2.39 *Monthly measurements (high and low tide average) of percent oxygen saturation at the Adams Point station from July, 1988 to June, 1996.*

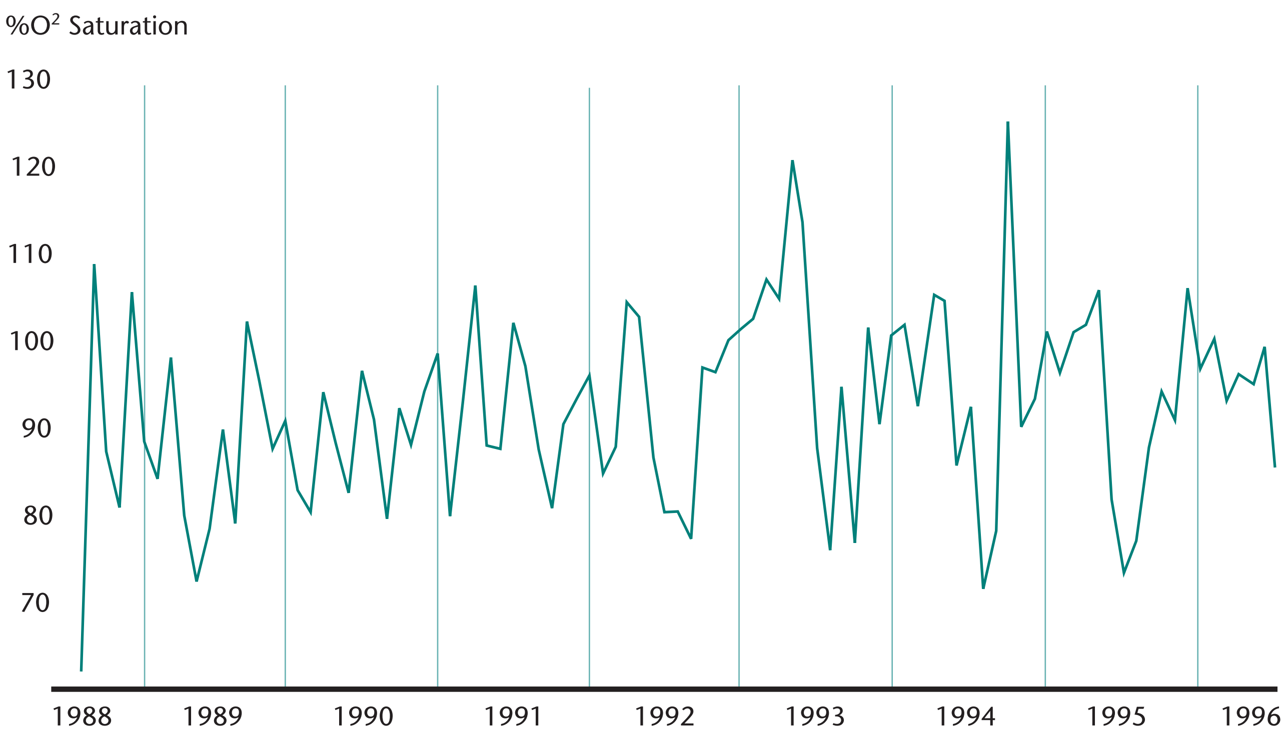


70% to 120%, depending on the time of day, tidal condition, and time of year (Figure 2.40).

In a three year project designed to assess the effect of stormwater runoff on contaminants in tributaries to Great Bay, measurements of dissolved oxygen were made in the freshwater portions of the tributaries and in the mouths of the tidal portions (Jones and Langan, 1994a, 1995a, 1996a). Data from this study indicates that dissolved oxygen in the freshwater portions of the rivers can get quite low, particularly at times of low flow. Freshwater measurements of D.O. often failed to meet the New Hampshire water quality criteria (WQC) of 75% saturation. Saturation in the tidal sites was generally 70% to 100% with few NH WQC violations. Though the water quality problems in the freshwater portions of the river may be related to eutrophication, it is likely that the summer low flow conditions result in stagnant conditions in the impoundments above the dams and that the sediment oxygen demand as well as respiration exceeds the oxygen repletion rates in water with poor rate of exchange. This condition is also acknowledged in the New Hampshire WQC, which includes a statement that WQC be met, "...except as naturally occurs". The low dissolved oxygen conditions measured in point samples in the Exeter River was verified in the summer of 1995 using a continuous datalogger. In August, 1995, dissolved oxygen ranged from 3 to 4 mg/L and 35% to 60% saturation. It should be noted however, that the summer of 1995 set a record for low rainfall and that the section of the river where the instrument was deployed was completely stagnant for weeks. Autumn storms, which produced increased flow, improved oxygen saturation to 80% by late October.

A study conducted by the Maine DEP (Mitnik and Valleau, 1996; Mitnik, 1994) measured dissolved oxygen at a series of stations in the freshwater and tidal portions of the Salmon Falls Rivers. These studies were conducted during the summers of 1993 and 1995, both of which were extremely dry. Depressed oxygen

Monthly measurements (high and low tide average) of percent oxygen saturation at the Squamscott River station from July, 1988 to June, 1996. **FIGURE 2.40**



conditions were detected at several stations in the freshwater portion of the river and near the bottom of a deep site (Hamilton House) in the upper tidal portion of the river. In 1959, average D.O. was less than 6 mg/l at sites along the lower seven miles of the freshwater portion of the river, with minimum values of 0 mg/l, and much higher levels in tidal and upstream freshwater sections of the river (NHWPC, 1960). In the the Maine DEP studies, the remaining stations in the tidal portion of the Salmon Falls River and in the Piscataqua River ranged from 80%-100% saturation at all depths. At the tidal site near Hamilton House in South Berwick, ME, the surface D.O. was usually near 100% saturation while the 5 meter depth D.O. was frequently below 50% saturation and was actually anoxic on one occasion in August. The low dissolved oxygen in the Salmon Falls River was attributed to eutrophication (intense plankton blooms) in the freshwater portion of the river, sediment oxygen demand (in deeper water) and stagnation caused by the series of impoundments on the river and extremely low flow conditions. The eutrophic conditions were attributed to excessive phosphorus from the four sewage treatment plants discharging to the river. An experimental phosphorus limitation period in 1995 resulted in significant reduction in phytoplankton in the impoundments. Based on recommendations from the Maine DEP study, upgrades of WWTFs in Berwick, ME, South Berwick, ME, Rollinsford, NH Milton, NH and Somersworth, NH are required to limit phosphorus discharges to the Salmon Falls River over the next few years.

Based on the existing data, it can be summarized that, in general, the Great Bay Estuary does not exhibit low dissolved oxygen conditions in the tidal waters. Even the shallow upper tidal reaches of the rivers exceed 5 mg/L in worst case scenarios (early morning low tides in mid to late summer), with an occasional measurement between 4.5 and 5 mg/L. It should be noted, however, that at some of these sites the periodic drops in oxygen at low tide in early morning may be a natural phenomena, particularly in very shallow water near marshes (Stanley and Nixon, 1992; Stokesbury et al., 1996). The warm temperatures and rich organic sediments result in high benthic respiration rates and could potentially draw down water column oxygen. The duration and spatial distribution of hypoxic effects are of greater importance with respect to biological effects than the instantaneous measurement of the level of dissolved oxygen (Stokesbury et al., 1996). Continuous attainment of the WQC for dissolved oxygen set by Maine DEP (85% saturation) and New Hampshire (75%) may be unrealistic and not achievable in certain water bodies, even in undisturbed estuarine systems. Perhaps a tiered approach similar to the Falmouth, MA nitrogen concentration standards would be appropriate.

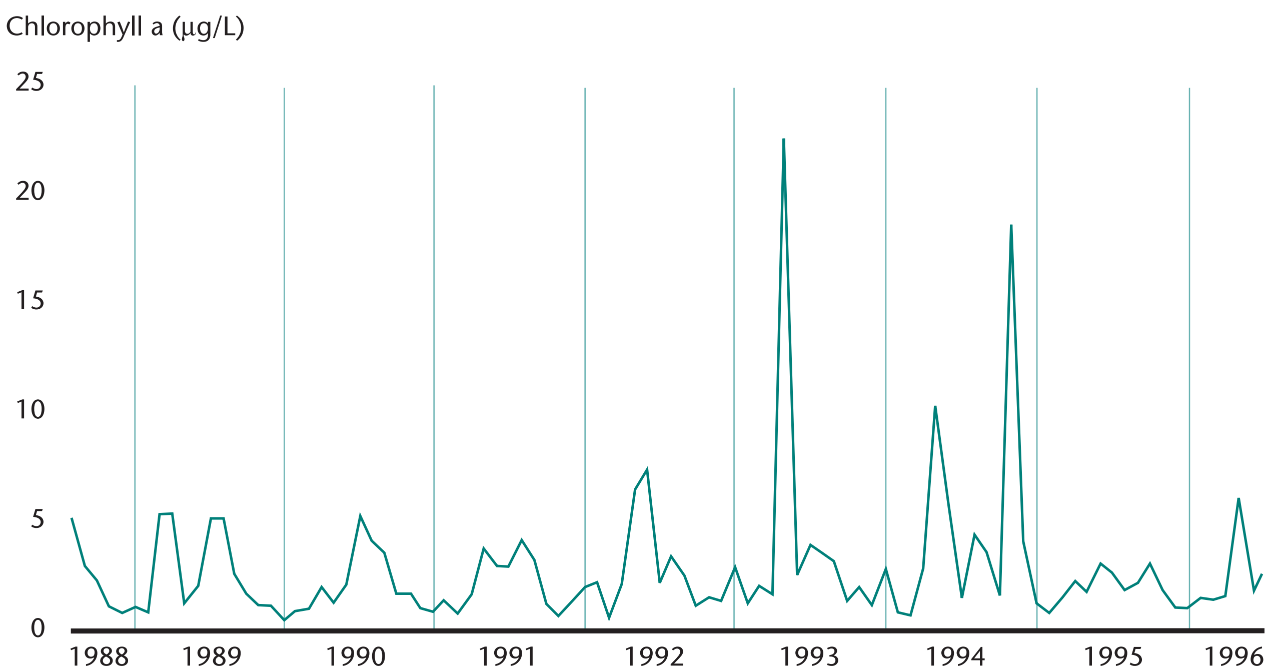
A review of available data does indicate, however, that the freshwater portions of some of the rivers (Salmon Falls, Exeter) can experience low dissolved oxygen episodes, and often for periods of up to several weeks during very low flow conditions in the summer. For the Salmon Falls River, the low dissolved oxygen can be attributed to excess nutrient input from WWTFs exacerbated by stagnant, impounded waters (Mitnik and Valleau, 1996; Mitnik 1994; Jones and Langan 1994a, 1995a, 1996a). It is unknown if there are present biological impacts associated with the low dissolved oxygen conditions in the freshwater impoundments. Historically, the existence of stretches of downstream, freshwater portions of the river being "devoid of fish due to lack of oxygen" was noted in the report by NHWPC (1960).

As is the case with nutrient data, there is considerably less data on dissolved oxygen in the Hampton/Seabrook Estuary than in Great Bay. As part of the Seabrook Station Environmental Studies Program, Normandeau Associates, Inc. has maintained a long term record of surface and bottom dissolved oxygen at a site outside the Harbor, but none in the estuary itself. The study of the potential of groundwater and surface water impacts from on-site sewage disposal systems described in an earlier section (Jones et al., 1996) was extended to include measurements in the summer of 1996 of dissolved oxygen in a number of small freshwater streams, marsh creeks, larger tributaries and in the Harbor itself (Jones, 1997). Out of a total of 139 samples taken in tidal streams and small marsh creeks from July, 1996 to June, 1997, seven D.O. measurements below 5 mg/l were recorded, all at low tide during the summer and early fall early in the day in small tidal creeks. All of the forty-seven measurements in the larger tributaries and in the Harbor itself were > 5 mg/l and generally greater than 75% saturation. Although the dataset is limited, it indicates that there are no low dissolved oxygen conditions that could result in biological impact in the Hampton/Seabrook Estuary.

2.4.5.2 Phytoplankton Blooms

The timing and intensity of phytoplankton blooms (as measured by water column chlorophyll) varies spatially in the Great Bay Estuary. Blooms in Great Bay and Little Bay generally occur in spring and fall, with variation between these two seasons as to when peak concentrations occur. Summer concentrations are generally lower than these peaks due to grazing, but are higher than winter concentrations. Peak concentrations at Furber Strait can reach as high as 20 µg/l (on one occasion in 1993 and one in 1994) but are usually on the order of 5-10 µg/l. Figure 2.41 represents chlorophyll concentrations averaged for high and low tides at the Furber Strait site. The average annual chlorophyll concentrations have ranged from < 2µg/l to > 3.5 µg/L with an eight year mean concentration of 3.2 µg/l. Chlorophyll concentrations in the lower estuary have a similar seasonal pattern (Langan, 1994), with blooms occurring in spring and fall. However, the peak concentrations are lower than in Great Bay, rarely exceeding 3 µg/l. Continuous measurements of chlorophyll were made on flood tide and ebb tide cruises in July, 1992, from the mouth of the harbor to the railroad bridge on the Squamscott River (Chadwick et al., 1993). On the flood tide, chlorophyll concentrations ranged from 1 to 1.5 µg/l from the harbor mouth to Dover Point; 2.5 to 3 µg/l in the upper Piscataqua River; 2-3 µg/l in lower Little

Monthly measurements (high and low tide average) of chlorophyll a at the Adams Point station from July, 1988 to June, 1996. **FIGURE 2.41**



Bay and 3-3.5 µg/l through upper Little Bay and Great Bay. Concentrations were slightly higher in some areas during the ebb tide cruise, however, the range of 1-3.5 µg/l was similar.

Peak concentrations in the tidal rivers follow a different pattern than areas in Great Bay, Little Bay and the lower Piscataqua River. Rather than a distinct spring bloom, chlorophyll concentrations gradually increase through the spring, and peak concentrations occur at some point from August through October. In the Squamscott River, peak concentrations for the period 1988 through 1996 were ≈ 30 µg/l, however, the peak in August, 1994, was 80 µg/l. The later blooms in the rivers are probably due to light limitation (from higher turbidity) in the spring.

Spinney Creek, a salt pond in Eliot, Maine, is susceptible to intense phytoplankton blooms by nature of its limited exchange of water (long residence time) with the Piscataqua River and elevated temperatures. The blooms can occur at any time from spring through fall and, the fall blooms are often the most intense. In the fall of 1996, a bloom of the naked dinoflagellate *Protocentrum spp*. lasted for several weeks and caused mortalities in oysters (*Ostrea edulis*) being raised in the creek. The cause of the bloom was attributed to regeneration of nutrients from macrophyte decay and little to no water exchange.

Bloom conditions in the other tributaries are best illustrated by examining data collected as part of a three year project to assess the effect of stormwater runoff on contaminant concentrations (Jones and Langan, 1994a, 1995a, 1996a). Intense blooms were recorded for two consecutive days after a rainstorm that followed an extended dry period in September, 1995. Highest intensities were recorded in the freshwater and tidal portions of the Salmon Falls and Cocheco rivers, suggesting that there may be periodic intensive bloom conditions in the freshwater and upper tidal reaches of these Rivers. These data are confirmed by Maine DEP studies in the Salmon Falls River (Mitnik and Valleau, 1996; Mitnik, 1994) where intense blooms were recorded in the freshwater impoundments and spilled over into the upper tidal portion of the river. Impacts to the tidal portion of the river were limited to low D.O. in bottom waters in a deep hole (6 m) adjacent to the Hamilton House. The low D.O. in the surface waters (fresh) was attributed to the respiration from phytoplankton bloom (caused by excess phosphorus and nitrogen from point sources), high water temperatures and long residence time of the water in the impoundments due to very low flow conditions, while the low bottom water D.O. was attributed to sediment oxygen demand.

Chlorophyll data collected at Furber Strait from 1973 to 1981 was compared to the 1988-1996 dataset. Means for the two periods were very similar: 3.4 µg/l for the 1973-1981 period and 3.2 µg/l for the 1988-1996 period. Seasonal patterns were also similar, as were minimum values (0 µg/l). The maximum value for the earlier data was 14 µg/l, and 20 µg/l in the more recent dataset. This comparison indicates that there has been little or no change on water column chlorophyll concentration over the 22 year period at this site.

Phytoplankton primary productivity, as measured by chlorophyll concentration, has been measured for many years outside the Hampton/Seabrook Estuary (NAI, 1996), however, it has been only recently that chlorophyll has been measured at sites within the estuary. Jones et al. (1997) measured chlorophyll concentrations in a number of small freshwater streams, marsh creeks, larger tributaries and in the harbor itself beginning in July 1996. Peak chlorophyll concentrations in the summer were approximately 3 µg/l in the larger tidal rivers and in the Harbor, and up to 28 µg/l in the small tidal creeks. Concentrations at all sites dropped through the fall and winter. Additional samples have been collected as part of the New Hampshire Estuaries Program to provide an improved spatial and temporal representation of the chlorophyll concentrations in Hampton Harbor.

2.4.5.3 Eutrophication

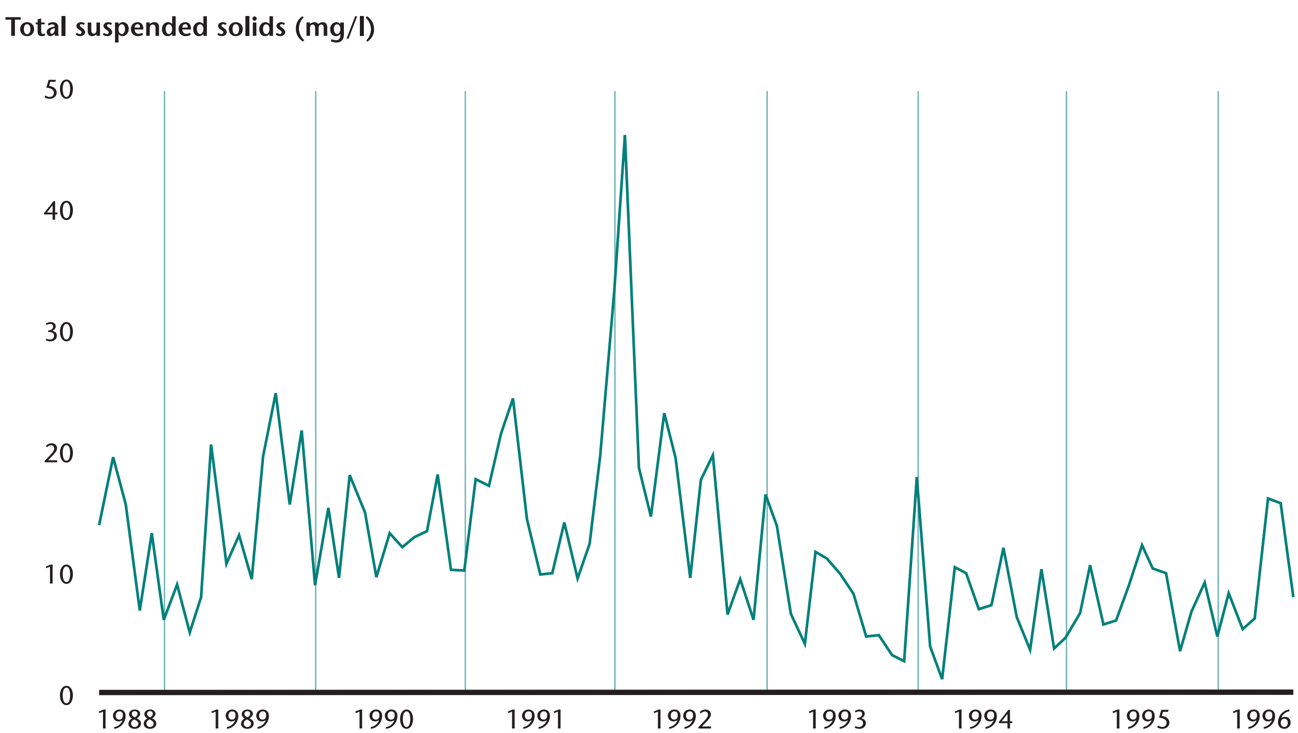
The Great Bay Estuary and other estuarine areas in New Hampshire had no cited incidences of eutrophic or hypoxic problems prior to 1985 (Whitledge, 1985). This report was a review of eutrophic or hypoxic estuaries nationwide, and more detailed New Hampshire information is provided below.

In addition to elevated nutrients, depressed dissolved oxygen conditions and phytoplankton blooms, other potential indicators of eutrophication include proliferation of opportunistic (green) macroalgae, reduction in water clarity, and loss of eelgrass. There has been some speculation that opportunistic macroalgal populations have increased in recent years (A. Mathieson, personal communication), however, this has not been substantiated with measured data. A project conducted during the summer of 1997 as part of the GBNERR monitoring program examined areal coverage and biomass of macroalgal species along an intertidal gradient for which an excellent baseline was established in 1973 (Chock and Mathieson, 1979). No changes in species, biomass and percent cover were documented (Langan and Jones, 1999).

Water clarity in the Great Bay Estuary is most affected by resuspension of fine grained sediments. Resuspension of sediments can result from human activities, such as dredging and boating in shallow water, however, natural causes, and in particular wind driven waves are the primary cause of resuspension (Anderson, 1974, 1975). Suspended sediments will be discussed in another section of this report, however it is useful to note here that at the two long-term monitoring sites in the Great Bay Estuary, suspended sediment concentration has decreased in recent years, and the annual mean is significantly lower at Furber Strait in the years 1993-1996 than from 1988 through 1992 (Figure 2.42).

Relative to eelgrass, a decline in the late 1980s in Great Bay attributed to the wasting disease, was followed by recovery in the 1990s. Areal coverage, density

Monthly measurements (high and low tide average) of suspended solids at the Adams Point station from July, 1988 to June, 1996. **FIGURE 2.42**



Rochester Wastewater Treatment Facility discharge

and biomass now exceed the early 1980s. Eelgrass has also been observed recently in areas where it has been absent for many years. It appears that eelgrass populations in the Great Bay Estuary are in good condition.

Based on the nutrient, dissolved oxygen and chlorophyll conditions, as well as the other potential indicators, there is no indication of system-wide eutrophication in the Great Bay Estuary, nor are there any documented trends that would indicate increasing nutrient enrichment. The physical characteristics of the estuary, including tidal height, relative flushing, a vertically mixed water column and high turbidity, in addition to the suite of parameters examined, would indicate that eutrophication in Great Bay is not an imminent problem. Though the data indicate that nitrogen may be limiting, light is also an important limiting factor due to resuspension of sediments and vigorous vertical mixing. There are indications, however, of potential problems in the freshwater portions of some of the tidal rivers and in the upper tidal reaches of the Salmon Falls and Cocheco rivers. Though both point and nonpoint sources may contribute to the problems observed there, low water flows and dams (impounded stagnant waters) contribute to water quality impacts. The location of a large point source on the Cocheco River (Rochester WWTF) and several smaller point sources (several WWTFs) on the Salmon Falls River are no doubt responsible for a large portion of anthropogenic nitrogen loading to these rivers. Though the potential for system-wide impacts from these rivers is remote, increasing the nitrogen load in the upper tidal reaches of these rivers could impact water quality in longer tidal stretches of both rivers, and potentially the upper Piscataqua River as well. Residence time is an important factor in determining sensitivity to nutrient overenrichment. For that reason, the tidal portions of the Lamprey and Squamscott rivers and areas in the southern portions of Great Bay would be considered areas susceptible to nutrient overenrichment since flushing times (complete water exchange) can be from two to three weeks for these areas in dry conditions. Therefore potential water quality impacts should be considered before this area is subjected to additional loading.

Based on the nutrient, chlorophyll and dissolved oxygen data reviewed, in addition to the lack of any indicators of eutrophication, there is no reason to believe that nutrient overenrichment is an issue in Hampton Harbor. Additionally, the rate of water exchange and short residence time of the water in the harbor would make it difficult for eutrophic conditions to develop in the estuary. With Seabrook-wide hook up to the new WWTF, future conditions are expected to be even better.

SUSPENDED SEDIMENTS AND TURBIDITY

Three review articles chronicle and synthesize most of the information available concerning suspended sediments and turbidity in the Great Bay Estuary. The Bibliography of the Geology of the Continental Shelf, Coastline and Estuaries of New Hampshire and Adjacent Regions (Ward and Pope, 1992) is a comprehensive report of all available literature up to 1992 concerning the geology and sedimentology of the New Hampshire region. An annotated bibliography for sediment based studies is included. A synthesis of the relevant research concerning the sedimentology (including the bottom and the water column) of Great Bay was presented by Ward (1992) and Short (1992). The most recent and up to date synthesis of research on suspended sediments and turbidity in the Great Bay Estuary is presented in A Monitoring Plan for the Great Bay National Estuarine Research Reserve: Final Report for the Period 07/01/95 through 06/30/96 (Langan and Jones, 1996). The synthesis of relevant research, annotated bibliography of relevant studies, and complete bibliography of known information presented here is based on these reports. Ward and Pope (1992) forms the basis of the complete bibliography up to 1992. The synthesis by Ward (1992) forms the framework for the review of existing information for suspended sediments and turbidity in the Great Bay Estuary. Where appropriate, segments of these reports are repeated here, as well as updated. Langan and Jones (1996), along within other recent reports, are used to update the synthesis and bibliographies.

2.5.1 SURFICIAL SEDIMENTS AROUND GREAT BAY ESTUARY

The surficial sediments in the Great Bay area have been strongly influenced by glacial advances and retreats during the Quaternary period (the last two or three million years of the Earth's history). During the last major glaciation (referred to as the Wisconsin), which began ~85,000 years ago and was at a maximum ~18,000 years ago (Flint, 1971), the large ice sheets removed much of the overlying soils and eroded the underlying bedrock (Chapman, 1974). Subsequently, extensive tills (unsorted sediments) and marine sands, silts and clays were deposited by the retreating glaciers (Delcore and Koteff, 1989). More recently, modern tidal flats, salt marshes and muddy to cobble beaches have developed adjacent to the estuary and its tributaries.

2.5.2 SHORELINE CHARACTERISTICS IN THE GREAT BAY ESTUARY

The intertidal shoreline of the Great Bay Estuary probably arrived close to its present day position a few thousand years ago when the rise of sea level slowed down. Since that time the estuary has been continuously modified by a slow sea level rise (presently about 1.5 mm/y, Hicks et al., 1983), wave effects, tidal action, biological processes, ice impact, and humans. Wave impacts in Great Bay Estuary are most important on the mudflat areas that often front the rocky or gravel shorelines (especially in the many embayments). Resuspension of fine-grained sediments from mudflats occurs during frequent wind events, increasing the turbidity of the nearshore and the overall estuary. These processes are discussed in more detail below. However, the wave energy is usually low and impact on the coarse-grained (gravel) beach sediments is probably small in many places.

Although no quantitative assessment of shore types has been done for the Great Bay Estuary (with the exclusion of the tidal marshes), qualitative observations based on aerial photographs and field observations have been made. Such studies indicate that exposed bedrock shorelines fronted by shingle beaches, small pocket beaches composed of sand to cobble size sediments, eroding till bluffs of little relief, muddy tidal flats, fringing marshes located on bedrock or coarse sediment, and large marshlands are all commonly found. Most frequently, the shoreline is exposed bedrock either fronted by cobble beaches, fringing marsh, relatively wide tidal flats, or large marshes. Large tidal flats dominate the intertidal and subtidal portions of Great and Little bays. Consequently, the surface area of the bays changes dramatically from high to low tide.

2.5.3 SOURCES OF SEDIMENTS

The sources of sediments for the intertidal and subtidal portions of Great Bay Estuary originate primarily from shore erosion, runoff from the watershed via inflowing rivers, and biological productivity. Erosion of the exposed bedrock surrounding much of the Bay provides irregularly shaped cobbles that form narrow shingle beaches. Some minor sandy beaches are located adjacent to eroding till deposits (e.g. Fox Point). Due to the rocky nature of the land surrounding the estuary and the relative thinness of the till deposits, it is unlikely substantial amounts of fine-grained sediment are contributed from shore erosion. Consequently, the source of new fine-grained sediments and turbidity is likely from freshwater tributaries. The impact of riverine inputs is most important following heavy rains which are more frequent in the spring. Jones and Langan (1996a) found the total suspended sediment concentrations in all the tributaries entering Great Bay following rain events to be higher than concentrations during dry periods, although the differences were less than 5 mg/l and usually not statistically significant. In addition, all of the associated rivers are dammed, reducing this potential source. The source of suspended sediments and turbidity on a day to day basis is more likely due to wind and tidal resuspension of the extensive subtidal and intertidal mudflats.

2.5.4 SUSPENDED SEDIMENTS

Spatially, the lowest suspended sediment concentrations occur in the lower estuary, while the highest generally occur in the upper estuary or within the tidal portions of the estuarine tributaries (Squamscott, Lamprey, Oyster, Bellamy, Cocheco, Salmon Falls or upper Piscataqua rivers). Ward (1994) measured the suspended sediment concentrations in the lower estuary (Portsmouth Harbor) and near the mid-estuary (Dover Point) over a number of tidal cycles in July, 1992. The concentrations were low and varied little across the channel and with depth in Portsmouth Harbor. The total suspended sediment concentrations ranged from 1.1 to 3.7 mg/l over a complete tidal cycle at the mouth of the Harbor and from 1.5 to 5.9 mg/l at a cross-section near Seavey Island. Similarly, Shevenell (1974) found suspended sediment concentrations were generally less than 3 mg/l at a station in the mouth of the Piscataqua River in 1972-1973, except during winter when concentrations exceeded 6 mg/l. According to Shevenell (1974), the main sources of particulate matter in the coastal shelf waters adjacent to the Piscataqua River were biological productivity, resuspension of bottom sediments and estuarine discharge from the Piscataqua River. Shevenell (1974) also noted particulate matter concentrations fluctuated seasonally and spatially due to meteorological effects (e.g., storms, high river discharges).

Total suspended sediment concentrations were higher in the mid-estuary, ranging from 2.4 to 12.7 mg/l over a tidal cycle at a cross-section at Dover Point in July, 1992 (Ward, 1994). The increase in total suspended sediments in the mid-estuary over the concentrations measured near the mouth reflects the impact of higher suspended sediment inputs from the upper estuary (e.g., Great Bay, upper Piscataqua River, tributaries).

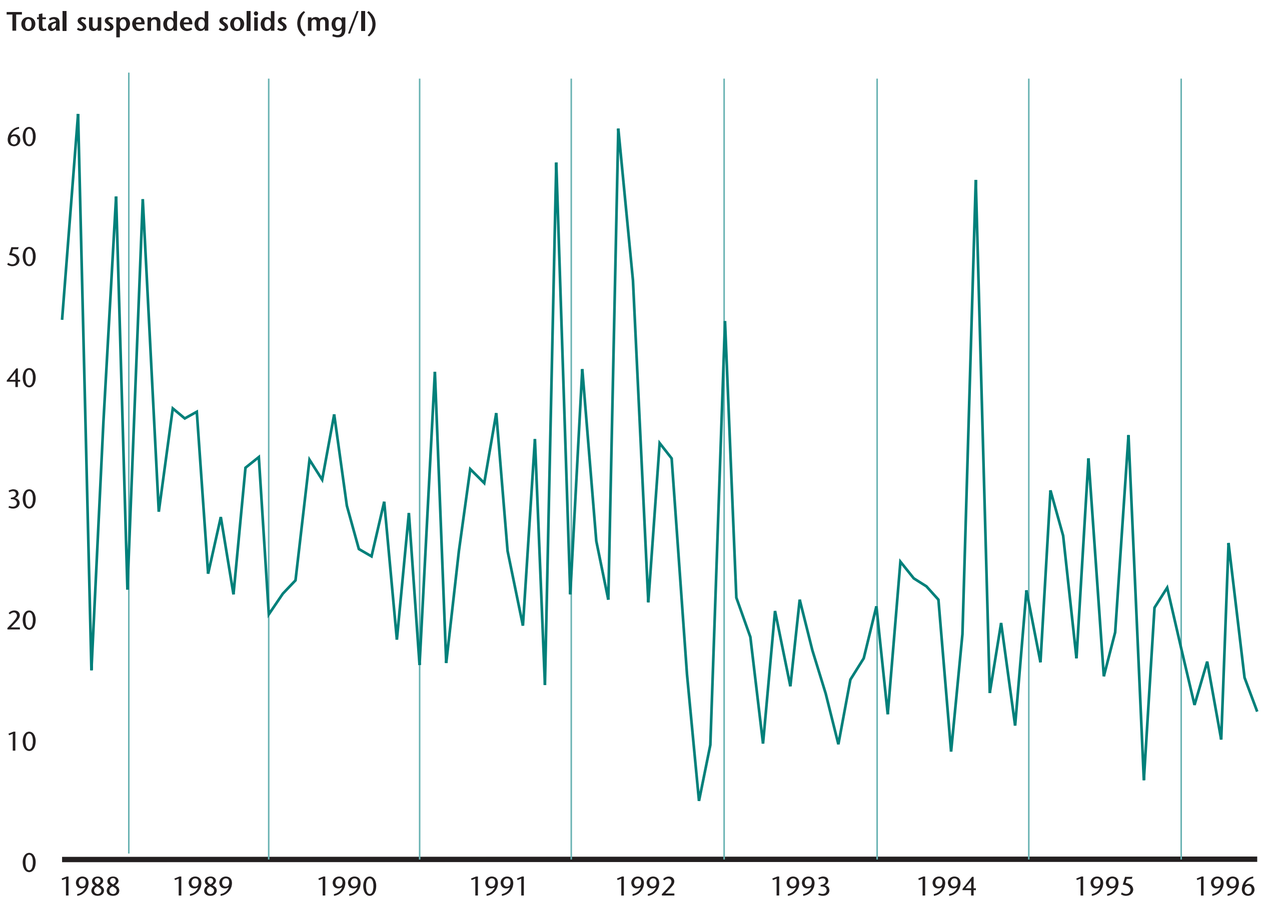
The spatial pattern of the total suspended sediment concentrations from the mouth of the estuary in Portsmouth to the upper estuary is reflected in the results of transects run in July, 1992 (Ward, 1994). The concentrations measured at ~high tide or early ebb ranged from 1.3 mg/l at the mouth to 17.7 mg/l at the entrance to the Squamscott River. Concentrations along the same transect run at ~ low tide and during the early flood ranged from 2.4 mg/l to over 50 mg/l at the Squamscott River.

Temporally, the highest concentrations occur in spring and fall, while summer and winter have lower concentrations (data from Loder et al. 1983, in Short, 1992). The total suspended sediment concentration off Furber Strait in the Great Bay averaged 11 mg/l from 1976 to 1978, with the lowest values in fall and winter. Unpublished data from Ward during 1991 to 1992 shows a similar pattern for Furber Strait. Short (1992) indicated the maximum suspended sediment concentrations occurred in the 1970s, although the averages are similar.

Langan and Jones (1996), focusing on the upper estuary, found that the suspended sediment concentrations from summer, 1995 to summer, 1996 were highest in the lower reaches of the Squamscott River (measured at Chapmans Landing) ranging from 5.8 to 42.7 mg/l and averaging 20.5 and 15.1 mg/l at low and high tide, respectively. The suspended sediment concentrations at Furber Strait ranged from 3.3 to 22.8 mg/l and averaged 9.8 and 7.5 mg/l at low and high tide, respectively. These averages are slightly lower than measured in the mid to late 1970s and in 1991/1992. Langan and Jones (1996) found the suspended solids concentrations at sites at Chapmans Landing and Furber Straits decreased from 1988 to 1996, significantly in some cases. Clear seasonal patterns were not apparent at these sites (Figures 2.42 and 2.43).

Lower concentrations for the 1995-1996 period were measured in the Lamprey River than in either the Squamscott River or at Furber Strait (Langan and Jones, 1996). Suspended sediment concentrations averaged 3.8 mg/l at both high and low tide in the Lamprey at the Town Landing. The suspended sediment

Monthly measurements (high and low tide average) of suspended solids at the Squamscott River station from July, 1988 to June, 1996. **FIGURE 2.43**



Adams Point in winter

concentrations in the Oyster River appeared to be similar to values measured for the Squamscott River (Jones and Langan, 1993a). Interestingly, there were no distinct differences on a seasonal bases in the Oyster River, nor were there consistent spatial variations. The average concentration in Oyster River were high, with a low tide mean of nearly 35 to 40 mg/l. However, this mean included samples taken in shallow water stations in the upper tidal reaches where local wind resuspension and other processes biased the results. The overall changes with time in the Great Bay Estuary need to be examined further.

The periodic nature of the suspended sediment load in the estuary has been described by Anderson (1970) who demonstrated large changes in concentrations over tidal cycles and over seasons. Suspended sediment concentrations ranged from ~2 to 18 mg/l in the channel at the entrance to the Bellamy River in Little Bay in response to tidal currents, resuspension events, spring discharge and ice effects. Large increases in the suspended sediment load can occur over tidal flats due to small amplitude waves (Anderson, 1972, 1973), extreme water temperatures caused by tidal flat exposure during summer months (Anderson, 1979; 1980), desiccation of the tidal flat (Anderson and Howell, 1984), rain impact (Shevenell, 1986; Shevenell and Anderson, 1985) and boat waves (Anderson, 1974; 1975). Webster (1991) investigated bedload transport on a tidal flat in Great Bay and found that the transport rates were related primarily to wind wave activity, although tidal currents may have enhanced movement. Webster (1991), also found that the benthic community appeared to affect bedload transport by disturbing the tidal flat surface (pellet mounds and feeding traces). Sediments resuspended along the shallow flats mixes with the channel waters, resulting in higher turbidity in the estuary. Thus, sedimentary processes which occur along the shallow flanks of the estuary have a large impact on the overall water quality.

2.5.5 SEDIMENTATION PROCESSES ON GREAT BAY TIDAL FLATS

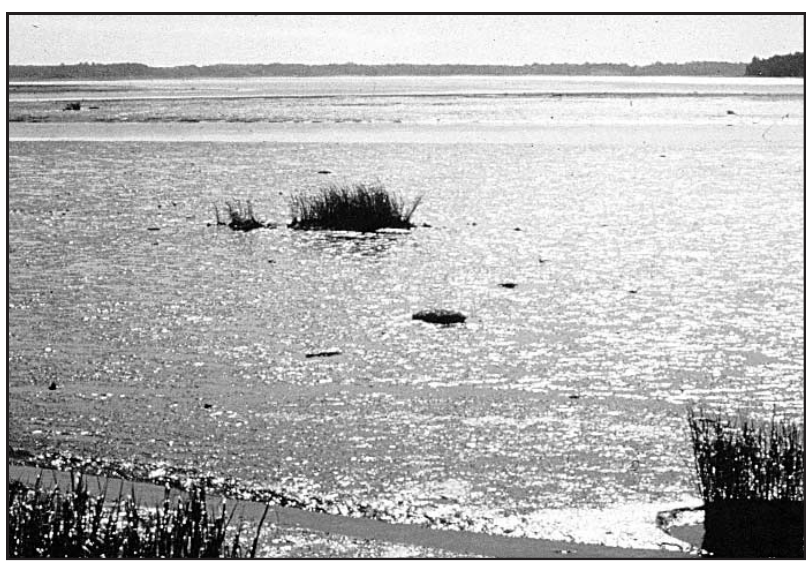
Anderson (1983) summarized the physical and biological processes influencing muddy intertidal flats, emphasizing the Great Bay. Anderson (1983) concluded that the main physical factors were: effects of ice, waves, sediment dewatering, mud and water temperatures, and rain. Biological factors included growth of benthic diatoms, algal mats, macrovegetation, bioturbation, pellet formation, biodeposition and changes in mudflat microrelief. Ice effects dominate in winter and early spring with breakup causing erosion. Wind resuspension was common much of the year. During summer, biologic processes dominate and deposition is more common. Storm activity in fall as biologic processes slow causes increased tidal flat erosion.

Wave action on the muddy intertidal flats causes erosion, resuspension, and subsequent transportation of the sediments. Tidal currents serve to distribute the sediments which are introduced via riverine sources, from bluff erosion, or from resuspension episodes on intertidal flats. In addition, strong tidal currents limit the seaward expansion of the tidal flats.

Sedimentation processes on the shallow tidal flats around the Great Bay are strongly influenced by biologic processes. Black (1980) found deposit feeders ingest muddy sediments, creating fecal pellets that behave hydraulically like fine-sand grains. Estimated feeding rates, for example, of *Macoma balthica* indicate the surface sediments are turned over 35 times per year (Black, 1980). Sickley (1989) demonstrated that tidal flat erosion was related to decreases in microbial populations and to the grazing activity of epibenthic macroorganisms. Sickley (1989) also showed suspended sediment concentrations to be related to benthic algal populations, which tend to bind the sediment.

Because of the temperate climate of the estuary, ice plays an important role in shaping the geomorphic and sedimentologic characteristics of the shoreline. During most winters much of the shoreline and intertidal regions of the bay are covered with ice. Ice tends to modify the shoreline by pushing sediments about and by forming gouges in the softer, muddy tidal flats. In winter during periods of ice movement, large amounts of sediment, clumps of marsh, and seaweeds are transported and eventually deposited elsewhere in the Bay (Mathieson et al., 1982; Hardwick-Witman, 1986; 1985; Short et al., 1986). Thompson (1975) found that ice on a tidal flat near Adams Point contained 0.58 to 27.2 grams of sediment per liter of ice. According to Thompson (1975), up to 50 cm of sediment was eroded from inner portions of the tidal flat, while up to 25 cm was deposited along the outer portion. Overall, the ice impact appeared to be erosional.

Suspended sediments have been measured in the Hampton/Seabrook Estuary as part of the 1994 Sanitary Survey (NHDHHS, 1994a), and was included in surface water sampling for studies on potential surface water contamination from septic systems (Jones, 1997). Samples have also been collected and analyzed from sites in the estuary as part of the monitoring supported by the NHEP. Total suspended solid concentrations in the Harbor are generally quite low, ranging from 1 to 6 mg/L, while in the smaller tidal creeks concentrations can be considerably higher, depending on tidal stage and wind speed and direction.



OTHER CONTAMINANTS OF POTENTIAL CONCERN

2.6.1 RADIONUCLIDES

The US EPA has published radiological surveys of the Portsmouth Naval Shipyard. Two of these documents have been obtained (USEPA, 1979; 1991). For both the 1977 and 1989 samples, materials from sites around Seavey Island and the Great Bay Estuary included sediments, sediment cores, biota and water. The 1977 study also included samples of vegetation and air samples. The results of both studies showed no evidence of radioactivity released as a result of Naval nuclear propulsion plant operations, based on cobalt-60 analyses. Detectable radioactivity in the biota and the environment surrounding the shipyard was attributed to naturally occurring isotopes or atmosphere-borne isotopes indicative of past nuclear weapons testing.

Seabrook Station has an extensive radiological monitoring program of the marine environment around Seabrook Station. The monitoring program includes sampling and radiological analysis of seawater, sediment, fish, lobster, mussels and algae in the area near Seabrook Station and the offshore cooling system discharge area, as well as control stations of similar environmental media collected in Ipswich Bay, Massachusetts. Continuous air samples are also collected at eight locations and direct radiation is measured at 42 locations around Seabrook Station. This is augmented by 16 additional direct radiation monitoring locations along the immediate Station fence line. All direct radiation monitoring locations include the use of six separate passive detectors. In addition, milk is collected from seven milk farms around Seabrook Station.

The program began in 1984, more than five years before Seabrook Station began operation. No radionuclides attributable to the operation of Seabrook Station have been detected. Naturally occurring radionuclides have been identified by the program including K-40, Be-7, Th-232 and its daughter products. Cesium-137 was detected in milk in very small quantities as the result of fallout from atmospheric nuclear weapons testing. The levels of radionuclides are consistent with those measured during the preoperational phase of the monitoring program. All analytical results are submitted to the U.S. Nuclear Regulatory Commission in the Annual Radiological Environmental Monitoring Report.

2.6.2 BIOTOXINS

Paralytic shellfish poisoning (PSP) was first recorded in 1972 in this portion of the Gulf of Maine (GOM). *Alexandrium* spp., blooms are probably transported south to New Hampshire coastal waters from a source population near the mouth of the Kennebec/Androscoggin rivers in Maine (Franks and Anderson, 1992). Local conditions may have some effect on blooms even though occurrences in NH are typically associated with large regional occurrences in ME & MA.

The NHDHHS, with support from NHF&G, conducts weekly sampling of mussels (*Mytilus edulis*) for PSP analyses at one site in Hampton Harbor. Since 1983, blooms have occurred during late spring to late summer. During 1983-89, the average weekly PSP levels were periodically >44 µg PSP/100 g tissue (the detection limit) & over the closure limit of 80 µg PSP/100 g tissue (NAI, 1996). Red tide blooms were reported to occur on a regular basis in 1989 (NHDES, 1989a), but only rarely since 1991 (NAI, 1996). PSP was detected at >44 µg PSP/100 g tissue in 1991, 1993 & 1994, but only during May-early June. PSP was detected at increasing concentrations on 3 consecutive occasions in May, 1995. Even though concentrations were below the closure limit, flats were closed because of the trend and some ME flats had already been closed. In 1996, there were no closures (NHDHHS, unpublished data). Concentrations of PSP remained at <44 µg/100 g mussel tissue from 4/1/96 to 10/27/96 in Hampton Harbor. Monitoring programs in both Maine and Massachusetts provide useful additional information. Little other information is available to document other harmful algal bloom events.

2.6.3 ACID RAIN

The NHDES has a database for acid rain at NH lakes and ponds (NHDES, 1996c). The results show an increase in pH in precipitation over the past 15 years from 4.0 to 4.3, and a significant increase in alkalinity over the past 15 years in some ponds. Even though most New Hampshire lakes showed no significant change in pH over the past 15 or 50 years, many lakes are still vulnerable to acid rain and have pH values of <6.0. No data are collected for tidal waters.

Acid deposition is primarily a result of emissions of nitrogen (NOx) and sulfur (SOx) oxides into the atmosphere. Monitoring of NOx has been conducted by the NHDES Air Resource Division at Manchester and Portsmouth since 1986, and SOx has been monitored at fourteen locations since the mid-1970s (NHCRP, 1997). Power generation produces 90% of SOx and 39% of NOx emissions in NH, while mobile sources produce 51% of the NOx. National Ambient Air Quality Standards are 80 µg/m3 for SO_2 and 53 ppb for NO_2. The annual mean concentrations for these two gases have decreased since 1990, from 10.63 to 18.58 µg/m3 for SO_2 and from 24 to 12 ppb for NO_2.

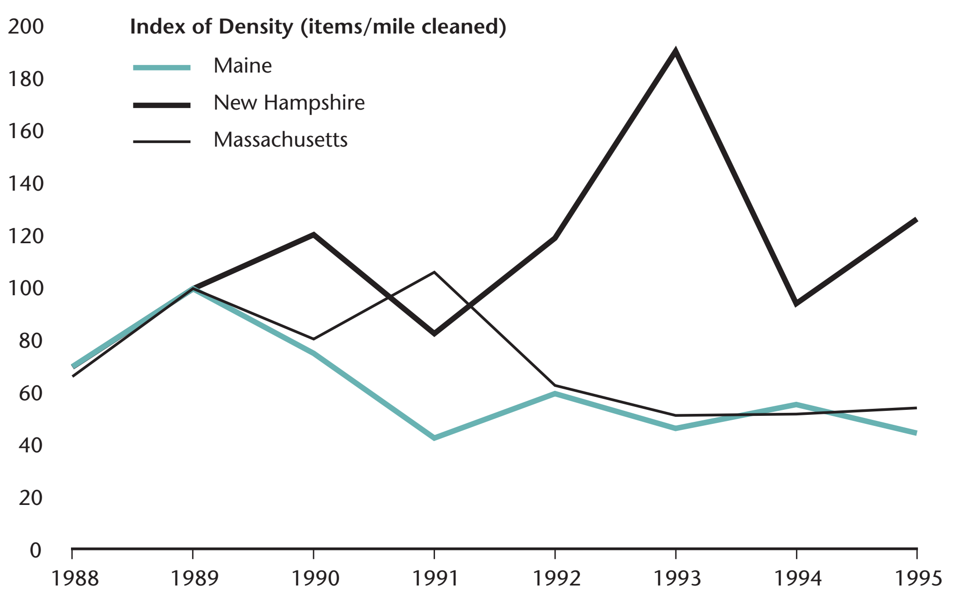
2.6.4 MARINE DEBRIS

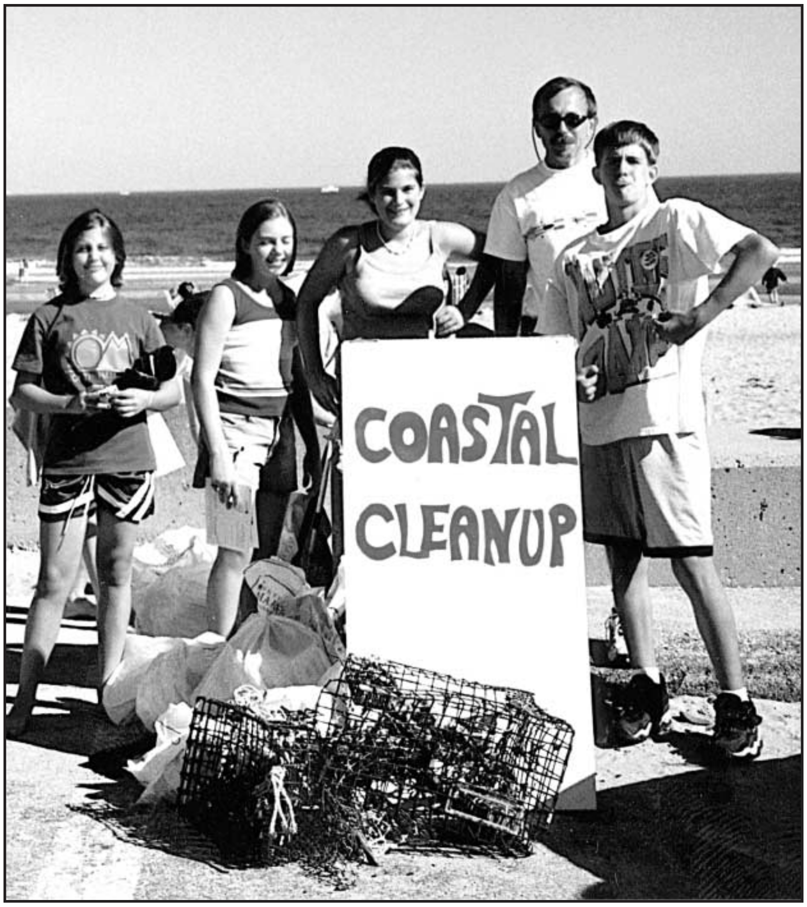
Data on marine debris clean up efforts since 1992 have been summarized by Salem High School (SHS, 1996). The information includes collection sites, numbers of debris items, type of debris, temporal trend analysis, and other data analyses. The New Hampshire clean up data are also analyzed in briefer fashion relative to the whole U.S. (Sheavly, 1996a) and international (Sheavly, 1996b) clean up efforts. The Piscataqua River Watershed Council is currently conducting a project with the Piscataqua Region Council on Marine Debris to reduce marine debris, especially bulk debris, through educational efforts (GOMC, 1997).

A recent review of historical marine debris distribution, temporal trends and sources of marine debris in the Gulf of Maine provides further analysis of data from New Hampshire, as well as identification of a range of policy approaches for addressing the issue (Hoagland and Kite-Powell, 1997). In general, it appears that New Hampshire, along with northern Massachusetts and parts of Nova Scotia, have relatively high densities of nearshore debris compared to Maine and southern Massachusetts. Since 1989, both

Index of bottles and associated items in marine debris from Maine, New Hampshire and Massachusetts, based on CMC data.

FIGURE 2.44





Maine and Massachusetts, which have bottle container laws, had slight reductions in beverage container debris while New Hampshire showed no reduction (Figure 2.44). Onshore sources of debris accounted for 80-85% of all debris, with much less coming from offshore sources (including commercial fishing gear).

2.6.5 OTHER CONTAMINANTS

The highest levels of ground-level ozone (O_3) in New Hampsire are in the Seacoast, where transport from large upwind urban areas is the greatest (NHCRP, 1997). The statewide average level, 0.047 ppm, has not changed much since 1990, and the range has been 0.45 to 0.5 ppm. The annual frequency of exceedences at individual locations has ranged from 0 in 1992 to 4 in 1991, with 3 in 1995.

Carbon monoxide (CO) is monitored in Manchester and Nashua. Levels appeared to improve during the 1990s. Air particulates have been monitored at 15 stations. From 1990-1995, none of them exceeded the standard. Particulate lead was monitored at 5 stations up to 1993, when monitoring ceased due to documented declines in response to removal of lead from gasoline.

Radon has been tested using home test kits since 1987. The action guideline is 4.0 pCi/l. Statewide, the geometric mean level is 2.8 pCi/l, and 36% of samples were > 3.9 pCi/l (NHCRP, 1997). The geometric means and percentage of samples > 3.9 pCi/l are 3.0 pCi/l and 38% for Rockingham County, and 3.6 pCi/l and 44% in Strafford County. Strafford County ranks second and Rockingham County is fourth amongst other state counties.

Data are kept on accidental chemical releases, which includes infectious agents, chemicals or radiological hazards. These usually occur at fixed sites or on roadways. The accidents usually involve release of petroleum products (77%) and toxic materials (15%). In 1993, Rockingham County had 138 events, the most of any county in the state, and Strafford County had 61. The statewide average from 1990 to 1994 was 373 events.

Chlorine is added to municipal drinking water (and WWTF effluent) as a necessary disinfection agent to kill possible microbial pathogens. However, the chlorine is highly reactive and can form potentially toxic chlorinated organic compounds, including chloroform, in the presence of naturally occurring organic compounds in water. The Maximum Contaminant Level (MCL) for chloroform is 5 µg/l. Chloroform was monitored in 12 municipal drinking water systems, including six in the coastal region, during 1995-1996 (NHCRP, 1997). The average chloroform concentration and risk (as number of excess cancers in one million people) were 44.2 µg/l and 3.17 cancers in Somersworth, 35.8 µg/l and 2.56 cancers in Exeter, 33 µg/l and 2.36 cancers in Portsmouth, 20.2 µg/l and 1.45 cancers in Rochester, and 17.7 µg/l and 1.28 cancers in Durham. All of these concentrations were greater than the MCL. The highest levels statewide were detected at Keene (49.8 µg/l), and Clairmont had the lowest levels (1.1 µg/l) and the only one under the MCL.

SUMMARY OF FINDINGS

The review of technical information on the status and trends for water quality in coastal New Hampshire showed a great deal of existing information for the different issues involved. Despite the abundance of information, much is still not understood and a number of issues are still significant. This section is a summary of what is known and what information gaps still exist.

FINDINGS

- There has been a general improvement in water quality in freshwater rivers and streams in coastal New Hampshire, in large part due to improvements in sewage treatment facilities. In 1996, all uses are fully supported in 100% of Coastal Basin and 99% of the Piscataqua River Basin streams and rivers.
- The water quality in the coastal shoreline and open ocean areas of the State's waters has improved to where they are also fully supporting all uses in 1996. Slower progress in estuarine waters, where uses are limited by numerous contaminants, has occurred.
- Fecal contamination levels have decreased in all coastal waters during the last decade as a result of improvements in wastewater treatment facilities.
- The spatial and temporal distribution of bacterial indicators in estuarine waters has been well documented in most areas. There are clearly sources of fecal contamination that persist in all areas of coastal New Hampshire.
- Fecal bacterial contamination is typically present at higher concentrations during low tide and after significant rainfall/runoff events.
- The major source of fecal contaminants in runoff is direct sewage contamination from leaky pipes and illicit connections in urban sewage pipe systems. These sources are also significant during dry weather.
- Other documented sources of fecal contamination include wastewater treatment facilities, septic systems, stormwater control systems and agricultural activities. Significant non-human sources of contamination other than from agricultural activities have not been documented.
- Recent sanitary surveys have expanded shellfish harvesting in areas with suitably low levels of fecal contamination.
- Indigenous bacterial pathogens, especially *Vibrio* spp., are present at relatively high levels in the Great Bay Estuary when water temperatures are warm.
- Tributaries to New Hampshire's estuaries have storm-related problems with trace metal contamination. Studies have shown how these contaminants have been transported, often in association with suspended sediments, throughout the downstream waters from tributaries.
- An historical database for sediment contaminants provides evidence for widespread contamination with trace metals and toxic organic compounds, and localized areas of high concentrations of these contaminants.
- Runoff from impervious surfaces is a significant source of both trace metal and toxic organic contaminants.
- Superfund sites located in close proximity to estuarine waters have had significant historical contamination and may continue to be sources affecting water quality.

- The large volume and trafficking of petroleum products through the Port of New Hampshire has resulted in numerous significant oil spills that have had directly adverse effects on estuarine biota.
- Atmospheric deposition of mercury is a significant concern in New Hampshire, while VOC emissions have been reduced.
- Models for predicting the fate of oil spills, trace metals and fecal contamination have been developed for numerous areas.
- Elevated tissue concentrations of toxic contaminants in estuarine biota have caused several consumption advisories. The relatively elevated levels of a number of contaminants is a critical concern.
- The highest levels of nitrogen and phosphorus occur in late fall through early spring throughout the Great Bay Estuary. The lowest levels occur in late spring through early fall.
- The highest levels of nutrients occur at the heads of tide in the tributaries, where sources such as upstream freshwater and WWTFs are most prevalent.
- Phosphate concentrations are usually low in freshwater, highest in upstream tidal rivers and low in Great Bay, Little Bay and Portsmouth Harbor.
- There is an inverse relationship between nitrogen concentration and salinity in Great Bay Estuary.
- Elevated nutrient levels occur in the tributaries of Hampton Harbor, but the concentrations in the Harbor itself are low. Conditions are expected to improve with the recently completed disconnection of septic systems in Seabrook.
- Current nitrogen concentrations, including annual means, seasonal patterns, and minimum and maximum concentrations, are similar to or lower than levels in the 1970s in most parts of the Great Bay Estuary and its tributaries. The exceptions are the freshwater portions of the Cocheco and Salmon Falls rivers, both of which are significantly impacted by WWTF effluent.
- Significant sources of nutrients include WWTFs, stormwater conduits, septic systems, lawns and golf courses, atmospheric deposition, natural organic debris and sediment recycling.
- Nitrogen loading from riverine sources is highest during late fall and early spring during times where rainfall events are more likely to cause runoff from land surfaces.
- The total nitrogen loaded to the Great Bay Estuary in 1996, based on some measurements and other estimations, was 718 tons. Nonpoint sources accounted for 48%, point sources 41% and atmospheric deposition 11% of the total. Similar contributions from different sources were determined for the Oyster River watershed.
- The estimated nitrogen loading, 718 tons/y, was slightly higher in 1996 than the NOAA estimate of 640 tons/y, published in 1990.
- Loading estimates for the Great Bay Estuary were below limits established for Buzzards Bay, MA.
- In general, the Great Bay Estuary does not exhibit low dissolved oxygen conditions in the tidal waters. D.O. can vary from 5 mg/l in summer during early morning low tides to 16 mg/l in winter.

- Areas in the Salmon Falls River can have exceptionally low D.O. and even anoxia, especially in the downstream freshwater and the upstream tidal portions during low flow periods in summer.
- Phytoplankton blooms in Great and Little bays can occur in spring and fall. Rather than experiencing distinct peaks, blooms in tidal rivers typically exhibit gradual increases in chlorophyll a concentrations with peaks in late summer or early fall.
- Intense bloom events have been observed in the Salmon Falls River coinciding with low D.O. conditions.
- There is no indication of system-wide eutrophication in the Great Bay and Hampton/Seabrook estuaries. Increased nutrient loading could cause problems in the upper tidal reaches of some of the tributary rivers.
- The major source of suspended sediments in the Great Bay Estuary is probably wind and tidal resuspension of subtidal and intertidal mudflat sediments.
- Paralytic shellfish poisoning levels have occasionally exceeded the closure limit of 80 µg PSP/100 g tissue in Hampton Harbor, the only monitoring site in New Hampshire. Little other information is available to document other harmful algal bloom events.

NEEDS

- With increasingly sophisticated monitoring and analytical methods being used, previously unidentified contaminants and sources are being detected. Thus, there is a continuing need to identify and eliminate sources of fecal and other contaminants that limit uses if coastal and estuarine waters.
- Establishment of a spatially comprehensive water quality monitoring program is needed to maintain existing harvestable shellfish areas and expand harvesting to new areas as management strategies to reduce contaminants are implemented.
- Continuing increases in human population and associated development, impervious surfaces and wastewater treatment demands will modify the capacities for watersheds to process contaminants. A better understanding of watershed factors and processes that affect the fate and transport of fecal and other contaminants is needed to frame effective strategies for managing transport of contaminants to surface waters.
- Studies on the occurrence of indigenous pathogens like *Vibrio* spp. and biotoxin-producing organisms would be useful for establishing baseline data and predicting potentially harmful conditions.
- A coordinated monitoring program that includes periodic analysis of sediments is needed to determine temporal trends for sediment contaminants. Monitoring for oil spills and atmospheric contaminants should be continued.
- Studies on the biological effects of single and multiple toxic contaminants are needed for some ‘hot spot’ areas of New Hampshire’s estuaries.
- With increasing human populations in the Seacoast, it is important to continue monitoring nutrient levels and dissolved oxygen, especially in the tidal river tributaries of the State’s estuaries.